

SEA TURTLE ENTRAPMENT AT A COASTAL POWER PLANT

Robert G. Ernest, R. Erik Martin, Bruce D. Peery and Douglas G. Strom, Applied Biology, Inc., P.O. Box 974, Jensen Beach, Florida 34958; J. Ross Wilcox, Florida Power & Light Company, P.O. Box 14000, Juno Beach, Florida 33408; and Nancy W. Walls, Applied Biology, Inc., 2968-A North Decatur Road, Decatur, Georgia 30033.

INTRODUCTION

Florida Power & Light Company (FPL) operates two 850-MW nuclear-fueled electric generating units on Hutchinson Island, St. Lucie County, Florida. Both units of the St. Lucie Plant collectively draw and discharge once-through condenser cooling water through an enclosed canal system connected with the Atlantic Ocean via submerged pipes. Although the offshore structures housing the intake pipes were designed to minimize entrainment of nektonic organisms, they apparently act as attractants to sea turtles. Upon entering the intake pipes, the turtles are rapidly transported into the canal system where they remain entrapped until manually captured and returned to the ocean.

Potential environmental issues relating to plant siting, design, construction and operation were addressed during permitting and licensing phases of plant development. In response to concerns raised by regulatory agencies, several design modifications were implemented. Additionally, baseline studies, including nesting surveys of the Hutchinson Island sea turtle rookery (Gallagher et al., 1972; Worth and Smith, 1976), were con-

ducted to establish background information on resident biotic communities. After a series of site certification hearings and satisfactory demonstration of environmental safeguards, the St. Lucie Plant received its operating license, and the first unit was placed on line.

The importance of the Hutchinson Island sea turtle rookery was clearly understood at the time of licensing negotiations (Routa, 1968). Pre-operational and operational monitoring focused on the potential disruption of nesting activities caused by plant construction and subsequent discharge of thermal effluents (O'Hara, 1980; Williams-Walls, et al., 1983; Proffitt, et al., 1986). However, the potential for entraining sea turtles with condenser cooling water was never realized until the plant began operating.

Soon after Unit 1 was placed on line, occasional sightings of sea turtles in the intake canal were reported. Concern for their safety prompted development of an efficient, non-injurious capture technique. Borrowing materials and experience from other researchers and local fishermen, a large-mesh "tangle" net was assembled and deployed. Immediate success in capturing turtles demonstrated its applicability.

As the scope and persistence of sea turtle entrainment at the St. Lucie Plant became apparent, a long-term canal capture and reporting program was developed in cooperation with the Florida Department of Natural Resources and the U.S. National Marine Fisheries Service. Capture methodologies evolved over the first several years as net materials, configurations and placement were varied in an effort to reduce entrapment times and

thereby minimize potential injuries and trauma to the turtles. Alternative capture techniques were also evaluated.

Methods of preventing sea turtles from entering the intake canal of the St. Lucie Plant were examined concurrently with the refinement of capture techniques. Engineering, legal and safety constraints precluded installation of physical barriers at the mouth of the offshore pipes. Consequently, laboratory studies concentrated on identifying effective deterrent mechanisms. To date, no practical deterrents have been identified.

The principal objective of the St. Lucie Plant canal capture program is to remove entrapped sea turtles from the intake canal and return them to the ocean as quickly and with as little stress as possible. However, in the process, a great deal of information relating to their biology and ecology has been obtained. As with most large, far-ranging marine animals, collecting data from live turtles in the field is both difficult and costly. Consequently, most researchers have relied on information obtained from adult nesting females, hatchlings, juvenile/sub-adults maintained under captive conditions and stranded individuals. By contrast, the St. Lucie Plant serves as a static offshore collection device, providing a continual supply of specimens of both sexes and a wide range of size classes. This unique opportunity to examine the local sea turtle population has been exploited through an extensive data collection program.

This paper describes the canal capture program currently in effect at the St. Lucie Plant and presents some of the data collected over a 10-year period beginning in 1976. Size-frequencies, recaptures and sex ratios are discussed relative to their implications to management policies affecting these threatened/endangered species.

MATERIALS AND METHODS

Plant Description - Hutchinson Island is one of a chain of barrier islands separating the shallow Indian River Lagoon from the Atlantic Ocean along Florida's lower east coast (Figure 1). The St. Lucie Plant is located on the island on a 437-hectare site midway between the Ft. Pierce and St. Lucie Inlets (27°21'N; 80°14'W). At this latitude, the continental shelf margin is approximately 33 km offshore. The Florida current flows roughly parallel to the margin but closer to shore (Gallagher and Hollinger, 1977), and a weak counter current is usually present near shore. Mean tidal range in the Atlantic Ocean in the vicinity of the plant is about 0.8 m. The adjacent sea bottom consists of shifting sand and shell rubble with occasional rock and reef outcroppings.

Condenser cooling water for the St. Lucie Plant is drawn from the Atlantic Ocean through two 3.7 m- and one 4.9 m-diameter pipes that rise vertically from the sea floor within separate reinforced concrete housings. These structures are located in approximately 7 m of water 365 m from shore (Figure 1).

Each of the intake housings is fitted with a 15.8 m-wide velocity cap. The caps, which are elevated 2.4 m above the mouth of the intake pipes, eliminate vertical draw. Maximum horizontal intake velocities at the caps are about 30.5 and 12.5 cm.sec⁻¹ for the largest and smallest pipes, respectively. At mean low water, the caps are 2.4 m below the surface.

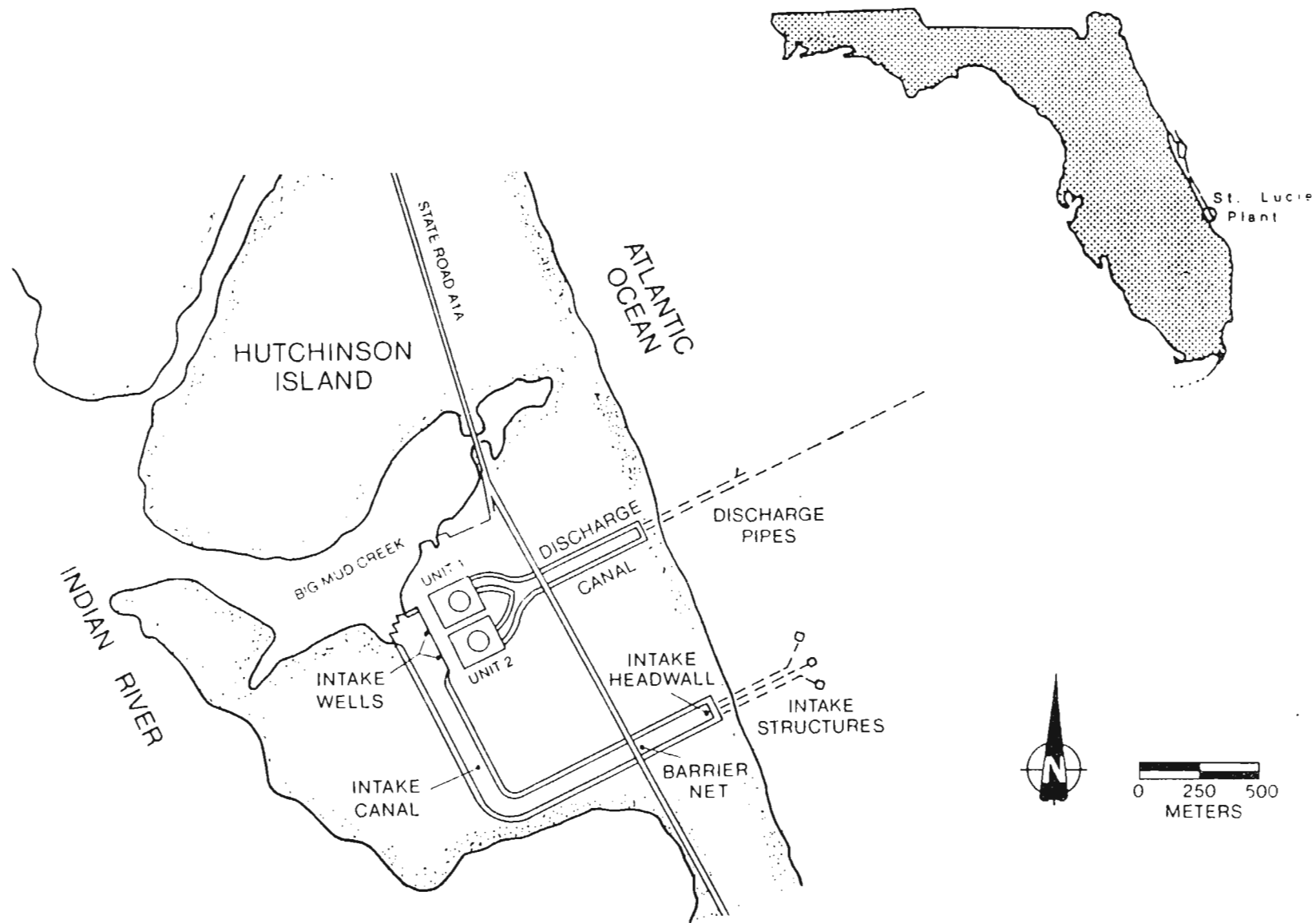


Figure 1. St. Lucie Plant site location.

Within the largest and smallest pipes, water velocities increase to 206 and 127 cm.sec⁻¹ respectively. The pipes travel shoreward beneath the sea floor, beach and dunes and terminate within two headwalls at the eastern end of a 1,500 m long, L-shaped intake canal (Figure 1). This canal transports cooling water to the plant. Water velocities within the canal are about 21 cm.sec⁻¹.

At the plant, cooling water is drawn from the bottom of eight separate intake wells (four for each unit). Water passes through a series of coarse grates, smaller mesh travelling screens and finally through the condenser cooling system. Heated water is discharged into a 670 m-long canal that leads to two buried discharge pipelines. These pass beneath the dunes and beach and along the ocean floor to submerged discharges, the first of which is approximately 365 m offshore and 730 m north of the intake structures.

Offshore construction of the Unit 1 intake and discharge systems took place between 1974 and 1976, with Unit 1 being placed on line in March 1976. Unit 2 intake and discharge lines were installed between 1981 and 1983, and Unit 2 became operational in May 1983. When operating at full capacity, each unit discharges approximately 32,350 l of heated effluent per second (738 million gallons per day).

Sea Turtle Capture and Data Collection - Turtles entrained with cooling water are generally confined to the easternmost segment of the intake canal between the headwalls and a barrier net located at the Highway A1A bridge (Figure 1). Water flow is low enough to permit turtles of all sizes to swim freely throughout this portion of the canal.

Over the years, nets used to capture turtles varied in length, were 2.7 to 3.7 m in depth and made of 40.6 cm stretch mesh, multi-strand nylon. Large floats were attached to the surface line and unweighted nylon rope was used on the bottom. The nets were usually deployed on Monday morning and retrieved Friday afternoon and were checked several times a day for captures. Turtles entangled in the nets floated at the water's surface until removed.

Larger turtles occasionally breached the A1A barrier net, and turtles smaller than 30.5 cm in carapace width could pass through its large mesh. Thus, as required, nets were also fished on the western side of A1A and in the vicinity of the plant's intake wells. Turtles not captured by nets were eventually removed from the canal at the plant's intake wells.

Captured turtles were identified to species, measured and weighed. Straight-line carapace length (SLCL) was measured from the precentral scute to the notch between the postcentral scutes (minimum carapace length of Pritchard et al., 1983). Curved carapace length, straight-line and curved carapace width, straight-line plastron length and tail length were also recorded. Adult turtles (SLCL >80 cm) were sexed based on relative tail lengths.

Beginning in 1982, blood samples were routinely collected and analyzed to determine the sex of immature turtles and to examine the relationship of hemoglobin values to the apparent physical condition of captured animals. Blood was removed from the paired dorsal cervical sinuses using the tech-

nique of Owens and Ruiz (1980). A single, sub-sample of whole blood was drawn and hemoglobin measured in grams per 100 ml by colorimetry using an A.O. 1010D hemoglobinometer. The remainder of the blood was then centrifuged for 15 minutes to separate cells and serum. Sex determinations were subsequently made by researchers at Texas A&M University using a radioimmunoassay for serum testosterone (Owens et al., 1978). Beginning in 1984, blood cell samples were provided to the National Marine Fisheries Service for the purpose of developing and refining methods for use in conducting turtle stock analyses.

Each turtle captured was examined for overall condition and subjectively ranked according to five relative condition categories based on weight, amount of activity exhibited, coverage of barnacles and/or leeches and occurrence and severity of wounds. Both systematic and occasional collections of fouling organisms (Frazier et al., 1985) were also made to identify potentially unique macrofaunal associations and to provide data relating to sea turtle migratory patterns.

Turtles were tagged with standard Monel or Inconel metal cattle ear tags on one or both front flippers and released back into the ocean at several locations on Hutchinson Island. Occasionally, turtles were provided to other researchers for use in behavioral studies.

RESULTS

Species Abundance - During the period from May 1976 through December 1985, a total of 1,310 sea turtle captures took place in the intake canal of the St. Lucie Plant (Table 1). Of the five species encountered, the loggerhead (Caretta caretta) was by far the most abundant (1,127 captures). Green turtles (Chelonia mydas) were the second most abundant (170 captures) followed by leatherbacks (Dermochelys coriacea; 7 captures), Kemp's ridleys (Lepidochelys kempi; 3 captures) and hawksbills (Eretmochelys imbricata; 3 captures).

Annual catches of loggerheads ranged from 33 in 1976, a partial year of plant operation, to 173 in 1979 (Table 1); the mean annual catch, excluding 1976, was 121.5. Green turtles were caught during every year of plant operation except 1976, but were most numerous in 1984 (69). The average annual catch of green turtles was 18.9.

Size-Class Distributions - Loggerheads removed from the intake canal ranged in length (SLCL) from 41.5 to 112.0 cm ($\bar{x} = 64.9 \pm 11.7$ cm) and in weight from 10.9 to 154.7 kg. Loggerheads as small as 74 cm standard SLCL (approximately 71 cm minimum SLCL; Henwood and Moulding, unpublished data) have been reported nesting along the east coast of Florida (Ehrhart, 1980). However, adults can only be reliably sexed on external morphological characteristics (e.g., relative tail length) after attaining a length of about 80 cm. Based on these divisions, data were segregated into three groups: juvenile/sub-adults (<70 cm; the demarcation between these two

Table 1

Total Number of Sea Turtles Captured Each Year in the St. Lucie Plant Intake Canal
May 1976 - December 1985

Species	1976	1977	1978	1979	1980	1981	1982	1983	1984	1985	Total
Loggerhead	33	80	138	173	116	62	101	119	148	157	1,127
Green	-	5	6	3	10	32	8	23	69	14	170
Leatherback	-	1	3	-	-	2	1	-	-	-	7
Hawksbill	-	-	1	-	-	-	-	-	1	1	3
Kemp's ridley	-	-	-	-	-	1	-	-	2	-	3
Total	33	86	148	176	126	97	110	142	220	172	1,310

components is not well established in the literature), adults (>80 cm) and transitional (70-80 cm). The latter group probably includes some mature and some immature individuals. Of the 969 individuals for which length data were collected, 76 percent were 70 cm or less in length, the majority of these measuring between 50 and 70 cm SLCL (Figure 2). About 11 percent of loggerhead captures involved adults, the remaining 13 percent representing individuals in the transitional size class.

Green turtles captured over the ten year study period ranged in length from 20 to 108 cm SLCL ($\bar{x} = 36.2 \pm 14.6$ cm) and in weight from 0.9 to 177.8 kg. Unlike loggerheads, all green turtles fell into either the juvenile/sub-adult or adult categories; individuals were either less than 66 cm or greater than 93 cm in length (Figure 3). The majority of greens captured (76 percent) involved individuals less than 40 cm in length and 9 kg in weight. Only five of the green turtles removed from the intake canal were adults.

Seasonal Distributions - On a seasonal basis, both loggerheads and green turtles occurred with the greatest frequency in winter. Thirty-four percent of all loggerhead captures occurred between January and March. However, when adult and juvenile/sub-adult segments of the population were analyzed separately, distinct differences in seasonal abundances were evident (Figure 4). Juvenile/sub-adults were most abundant during the winter, whereas adults were captured most often in summer; nearly 65 percent of all adult loggerhead canal captures occurred between June and September.

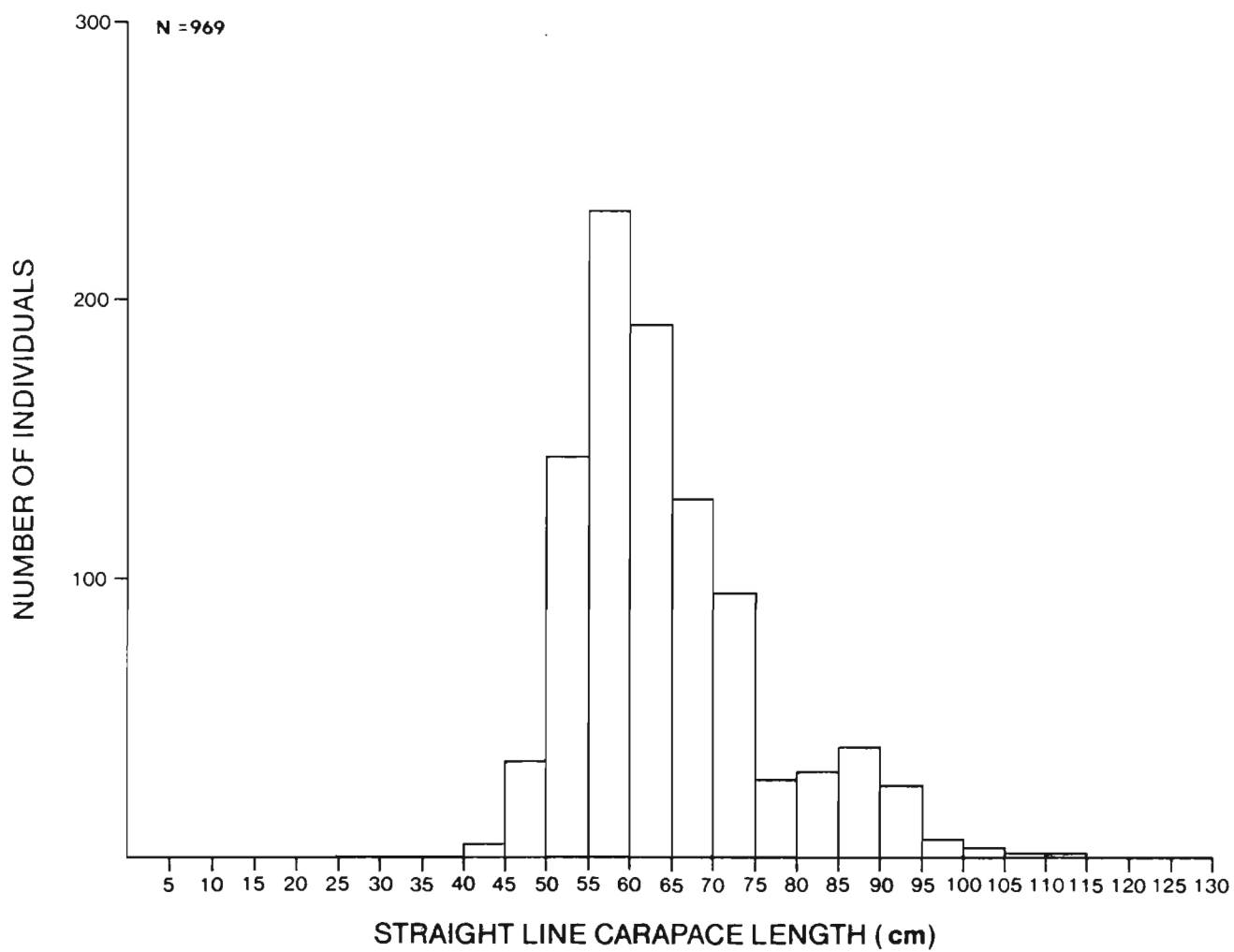


Figure 2. Size-class (SLCL) distribution of loggerhead turtles captured in the St. Lucie Plant intake canal, 1976-1985 (N=969).

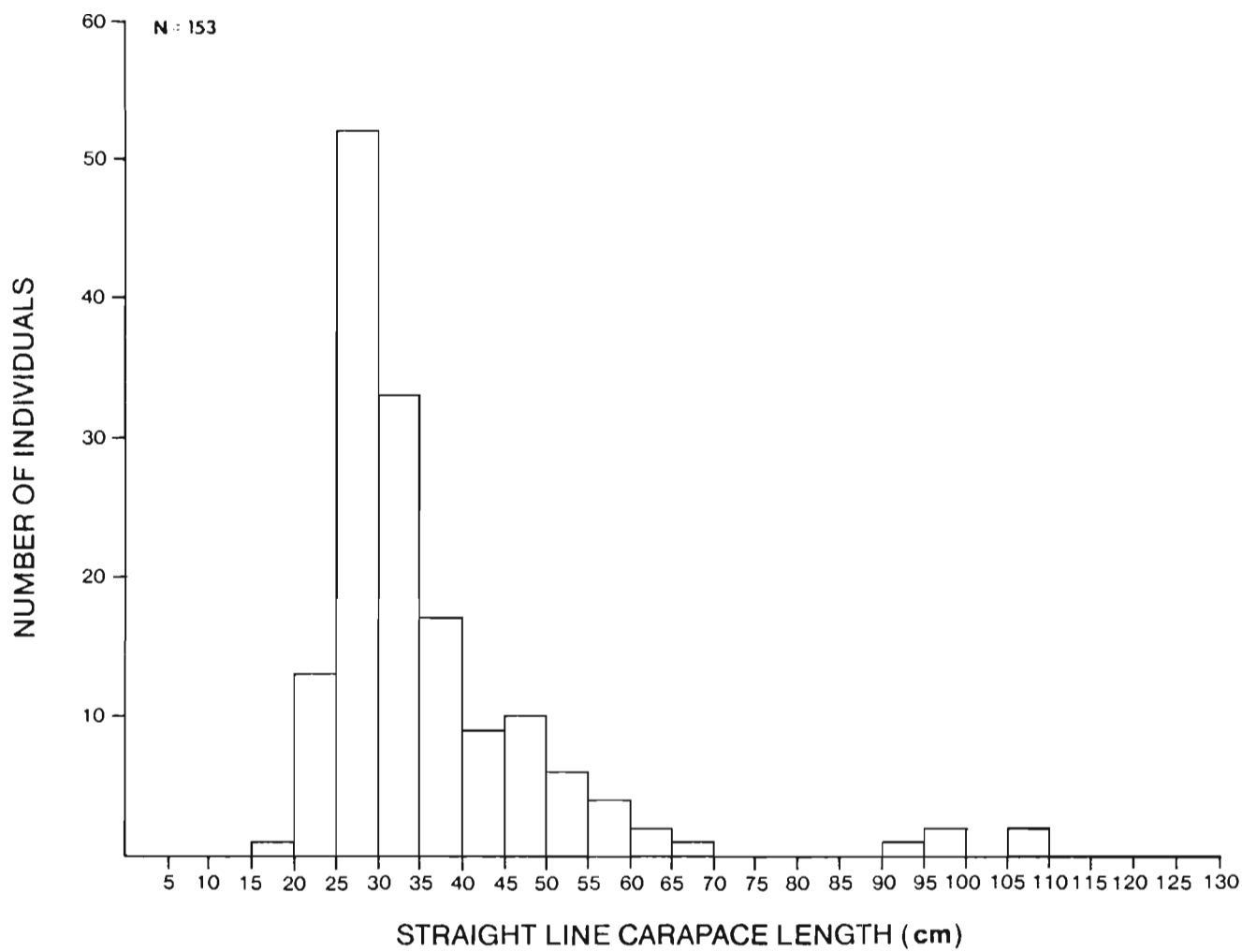


Figure 3. Size-class (SLCL) distribution of green turtles captured in the St. Lucie Plant intake canal, 1976-1985 (N=153).

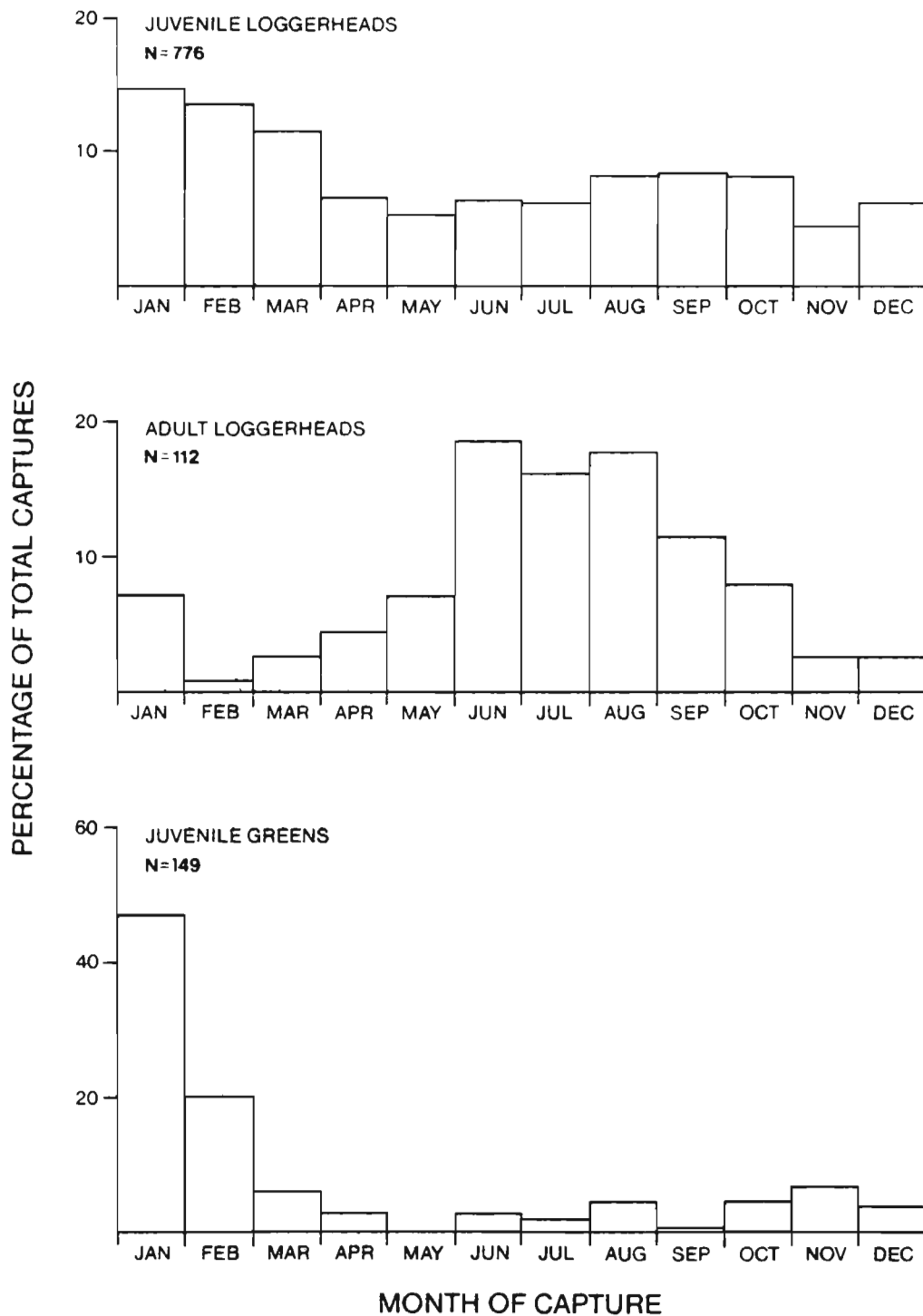


Figure 4. Percentage of all juvenile loggerheads (N=776), adult loggerheads (N=112) and juvenile greens (N=149) captured each month in the St. Lucie Plant intake canal, 1976-1985.

The seasonal distribution of green turtles was very skewed, 62 percent of all captures occurring during January and February (Figure 4). Catches of other species were scattered throughout the ten-year study period. All but one of the seven leatherbacks were collected in February, March and April. The hawksbills were captured in March, July and August and the Kemp's ridleys in January and February.

Sex ratios - Since 1982, 142 immature loggerheads (defined by Wibbels et al., 1984 as individuals less than 76 cm SLCL) were sexed by means of radioimmunoassay for blood testosterone. Females outnumbered males by a ratio of 2.38:1.00. This ratio is significantly skewed in favor of females (χ^2 , $P = 0.05$).

Recapture Frequencies - Of the 1,310 sea turtle captures recorded between 1976 and 1985, 54 represent turtles previously captured in the intake canal. A total of 32 loggerheads and one green were captured more than once, one loggerhead being captured on six different occasions. Recapture intervals for loggerheads range from 4 to 858 days with a mean of 126.7 days (s.d. \pm 151.3 days). About 60 percent of all recapture incidents occurred within 90 days of previous capture and 94 percent occurred within one year (Figure 5). The average period between first and last capture for loggerheads was 215.4 days (s.d. \pm 202.9 days). Over 80 percent of the recapture incidents involved juvenile/sub-adults.

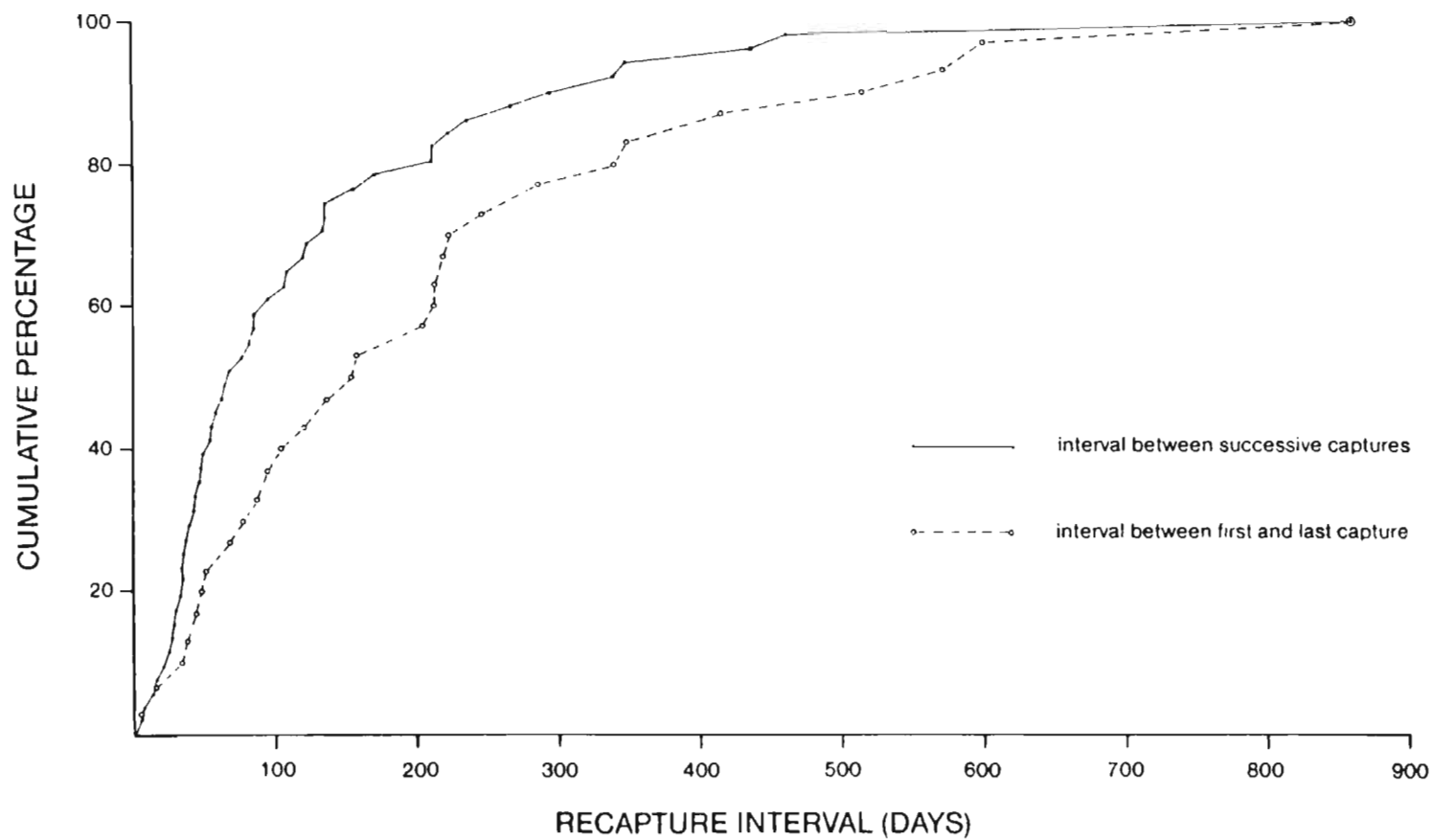


Figure 5. Cumulative percentage of all loggerhead recaptures occurring within various time intervals, St. Lucie Plant intake canal, 1976-1985.

DISCUSSION

Hutchinson Island supports a large nesting population of loggerheads and to a much lesser extent some nesting by greens and leatherbacks (Williams-Walls et al., 1983). During 1985, 4,877 loggerhead, 64 green and 18 leatherback nests were recorded along its 36.0 km shoreline (ABI, 1986). Canal capture data for the St. Lucie Plant indicate that the nearshore area adjacent to the plant also supports substantial numbers of juvenile/sub-adult loggerheads and greens. Periodic incursions by immature Kemp's ridleys, leatherbacks and hawksbills were also evidenced.

Nearshore coastal areas along the eastern seaboard of the United States apparently serve as migratory pathways and/or aggregation areas for juvenile and sub-adult loggerhead and green turtles. In Florida, relatively large numbers of loggerheads have been reported from the Port Canaveral ship channel (Ogren and McVea, 1982) and both species have been collected in the Mosquito and Indian River Lagoons (Ehrhart, 1983). Incidental catches of loggerheads in shrimp trawlers suggest similarly high abundances of immature animals off the coasts of Georgia and South Carolina (Hillestad et al., 1982). Although habitat utilization during different juvenile and sub-adult stages of the sea turtle life cycle is poorly understood, the importance of Atlantic coastal waters as a developmental habitat is becoming increasingly clear.

The degree to which canal capture data reflect the structure and dynamics of the population at large is primarily dependent on behavioral characteristics of each species. Only those individuals within a relatively limited nearshore area are susceptible to entrainment. However, for the purpose of this discussion, it is assumed that there is no intra-specific discriminant avoidance of the intake structures between sexes and/or size classes.

Size-class data for loggerheads entrapped in the St. Lucie Plant intake canal show a preponderance of juveniles/sub-adults within the "local population". Very similar size-frequency distributions have been reported for the Canaveral ship channel (Ogren and McVea, 1982), Mosquito/Indian River Lagoon (Mendonca and Ehrhart, 1982), and Georgia and South Carolina coastal waters (Hillestad et al., 1982). In none of the populations sampled did adults account for more than 15 percent of the total catch; 11 percent of the Hutchinson Island population was comprised of adults. Although it is possible that habitat segregation and/or gear selectivity may favor capture of juveniles/sub-adults over adults, the similarity of three geographically discrete data sets suggests a relatively uniform population structure throughout inshore coastal waters of the southeastern United States.

Also strikingly similar among east coast loggerhead "populations" is the lack of individuals smaller than 40 cm in carapace length. Carr (1986) recently elaborated on an ecologic mechanism to explain the "lost year" in loggerheads, a period during which post-hatchling turtles are rarely

encountered in coastal waters. Nurtured by food supplies concentrated on the surface by oceanic fronts and convergences, the small loggerheads leaving natal beaches may drift for four or five years in the currents and eddies of the North Atlantic gyre before finally leaving the pelagic environment to adopt a benthic mode of existence. Data from the Azores, in the eastern Atlantic, and from the east coast of Florida, including 10 years of canal capture data from the St. Lucie Plant, are consistent with this hypothesis.

By comparison, the majority of green turtles captured in the St. Lucie Plant intake canal was less than 40 cm in length, suggesting that greens depart the pelagic habitat at a smaller size than loggerheads. The canal capture data also indicate that green turtles appear in south Florida coastal waters at a smaller size than has been documented for most other coastal areas. In Hawaii and Australia, juvenile greens are not seen inshore until they reach a size of about 35 cm (Balazs, 1982; Limpus and Reed, 1985). Likewise, of the hundreds of Chelonia examined by Carr and Caldwell (1956) in the Cedar Key-Crystal River region of west Florida, the smallest was 13.5 inches (34.3 cm) long. Only in the Virgin Islands are there accounts of turtles less than 30 cm occurring with regularity in coastal waters. Schmidt (1916) reported that individuals between 23 and 30 cm in length were common there during January, February and March, the same period during which the majority of small greens are taken at the St. Lucie Plant. Whether animals of this size range are actively leaving the pelagic environment or are physically being forced from it by winter disturbances remains to be determined. Because much of the historical information

relating to the population structure of green turtles has been derived from turtles captured by net, gear selectivity cannot be overlooked when accounting for the "absence" of small turtles in areas otherwise known to support juveniles and sub-adults.

There is relatively little published information regarding green turtles inhabiting coastal waters of the southeastern United States. The St. Lucie Plant canal capture data and previous published reports of turtles in the Mosquito/Indian River Lagoon allow comparison of green turtle "populations" utilizing two distinct environments within this coastal zone.

The average size of the Mosquito/Indian River Lagoon green turtles was 48.2 cm, with 60 percent of all individuals measuring between 50 and 70 cm and only one percent measuring less than 30 cm (Mendonca and Ehrhart, 1982; Ehrhart, 1983). By comparison, green turtles captured at the St. Lucie Plant averaged 36.2 cm SLCL with less than nine percent measuring between 50 and 70 cm and over 40 percent measuring less than 30 cm. In both areas, animals in the 75-90 cm size range were absent. Some adults larger than 90 cm were captured at the St. Lucie Plant, whereas no adults were taken in the Mosquito/Indian River Lagoon. The absence of adults and the generally larger size of the Mosquito/Indian River Lagoon green turtles compared with those from the St. Lucie Plant suggests distinctly different population structures between the two areas.

Mendonca and Ehrhart (1982) provided strong evidence to indicate that green turtles use Florida's coastal bays and lagoons as development habitats, foraging on the seagrasses that constitute their principal diet (Mortimer, 1982; Mendonca, 1983, Bjorndal, 1985). Historically, these inland waters supported large populations of green turtles (Ehrhart, 1983). As juveniles, green turtles experience a transition in diet from omnivory to herbivory (Ernst and Barbour, 1972; Carr, 1980; Davenport and Oxford, 1984). Until this transition is complete, there is probably no compelling need for small greens to enter inland waters. Consequently, following the pelagic stage, many of these smaller animals may spend an extended period in the nearshore coastal environment. This would account for the preponderance of small turtles off Hutchinson Island. It would follow that as an herbivorous diet becomes increasingly more important to their existence, they eventually migrate inland where attached vegetation is much more abundant.

Mendonca and Ehrhart (1982) suggested that as the green turtles within the Mosquito/Indian River Lagoon approach maturity, they begin to leave the lagoon. The absence of turtles in the 70-90 cm size classes off Hutchinson Island indicates that this transitional group may leave Florida coastal waters all together. It is likely that this size range represents individuals being recruited back into the parent population to begin alternate migrations between traditional breeding and foraging grounds (Meylan, 1982). Whether these same animals will eventually constitute all or part of the Florida nesting population is unknown.

The heavily skewed seasonal distribution of green turtles off Hutchinson Island probably reflects measured movements of this species in response to changing environmental conditions and/or changes in environmental requirements (Carr, 1980). Both geographical and local movements are likely to increase as water temperatures begin to decline (Mendonca, 1983). Additionally, northeasterly storms may extirpate some "lost year" turtles from the pelagic environment, thus increasing the number of small turtles found near shore during the winter. Much additional information will be required to determine the relative contribution of each of these factors to seasonal abundance patterns of green turtles at the St. Lucie Plant.

Although not as extreme as that observed for green turtles, the seasonal distribution for juvenile/sub-adult loggerhead turtles was also skewed toward colder months of the year; 40 percent of all captures occurred between January and March. By contrast the most productive capture period for loggerheads in the Mosquito/Indian River Lagoon was between April and October when highest water temperatures prevailed (Mendonca and Ehrhart, 1982). This may suggest seasonal migrations of loggerheads between inland and coastal waters and would account for the relatively brief residence time observed for Caretta in the Mosquito/Indian River Lagoon.

Adult loggerheads, although present throughout the year, were most abundant during the summer. May through August represents the nesting season for loggerheads on Hutchinson Island, with peak nesting occurring in June and July (Williams-Walls et al., 1983). Females approaching the beach have a greater chance of encountering the intake structures than those

remaining offshore, and as the nesting activities of females increase, canal captures of adults increase accordingly.

Males and females were captured in nearly equal numbers between November and April, indicating some inshore/offshore or latitudinal movements on a regular basis. However, during the summer, females were captured with over six times the frequency of males. Although the timing and location of mating is not well documented, copulation is believed to occur off nesting beaches prior to the nesting season (Ernst and Barbour, 1972; Geldiay et al., 1982, Mrosovsky, 1983). St. Lucie Plant data suggest that either mating off Hutchinson Island is confined primarily to waters more distant from shore than the intake structures or that reproductively active males are less attracted to the structures than are females. Changes in behavioral characteristics of sexually active male turtles have been previously reported (Booth and Peters, 1972; Balazs, 1980).

Repetitive captures of individually marked turtles in the St. Lucie Plant intake canal provide some clues as to the length of residence within the Hutchinson Island population. Over the 10-year monitoring period 32 loggerheads and one green turtle were captured more than once. These turtles represent 2.6 percent of the 1,256 individuals removed from the canal during that period. Turtles were released at numerous locations along the island, and there was no clear evidence of association between site of release and probability of recapture. The longest period between first and last capture was less than 30 months, the average interval being a little over six months. Assuming no learned avoidance behavior to the

intake structures, the data suggest a relatively brief residence time in the Hutchinson Island population. Although tag retention may be a hindrance to monitoring individual turtles on a long-term basis (Carr, 1980; Hughes, 1982; Henwood, 1986), the general lack of individuals in the canal with tag scars indicate a true absence of long-time residents. Similar findings were made for both greens and loggerheads in the Mosquito/Indian River Lagoon where the longest observed residence times were 20 and 15 months, respectively (Mendonca and Ehrhart, 1982).

At least some of the loggerheads released from the canal are known to have migrated elsewhere. Two tagged loggerheads were recovered in North Carolina, one in South Carolina, one in Georgia and one near St. Augustine, Florida. Two others were found stranded along beaches just north of the study area. For the five tag returns received from the most distant locations, all involved juvenile/sub-adults (50-70 cm SLCL) released in January, February and March; all were recovered between July and December of the same year. Thus, some turtles not only leave the immediate area of the plant but travel considerable distances over relatively short time spans. Although no turtles released from the St. Lucie Plant have been recovered to the south, three sub-adult loggerheads tagged in the Canaveral ship channel were later captured at the plant.

Long-range movements of adult females nesting on the east coast of Florida have been reported (Meylan et al., 1983), but little published information exists for the juvenile/sub-adult segment of this "population". This is related primarily to the general inaccessibility of non-adult

turtles. Compared with studies of nesting females, relatively few research activities are directed toward the capture and tagging of individuals in the 40-70 cm size range, even though this is the most abundant component of the nearshore population. Most tag returns from immature turtles come from incidental captures in fishing gear and strandings on seasonally monitored nesting beaches. Consequently, tag returns from canal-captured turtles, although few in number, may reflect a substantial amount of movement within the juvenile/sub-adult segment of the western Atlantic loggerhead population. Because overall canal captures have not declined over the years of plant operation, it also appears that emigration from and immigration into the Hutchinson Island area is relatively well balanced.

Until recently, sex ratios of sea turtle populations were based primarily on counts of adults in the wild. Observed frequencies often varied considerably between locations and seasons (Ross, 1984) and sex-dependent behavioral characteristics made interpretation of ratios difficult (Balazs, 1980; Limpus and Reed, 1985). A relatively new radioimmunoassay technique now permits safe and easy sexing of juvenile/sub-adult animals (Owens and Ruiz, 1980). Because of their relative abundance in the Hutchinson Island population and their availability year around, juvenile/sub-adult loggerheads lend themselves nicely to studies of population sex ratios.

For juvenile/sub-adult loggerheads taken from the St. Lucie Plant intake canal, females significantly outnumbered males by a ratio of 2.4 to 1.0. Wibbels et al. (1984) similarly found that the sex ratios of immature loggerheads taken from the Canaveral ship channel were significantly skewed

in favor of females. These data taken collectively with information from the Indian River and the Chesapeake Bay prompted Wibbels et al. (1984) to hypothesize that a skewed but similar sex ratio exists throughout US coastal waters, and individuals within the region may constitute a single population. Various explanations for, and the adaptive significance of, a female-skewed population have been presented (Balazs, 1980; Mrosovsky, 1980), but at present, additional data from a number of different locations and over longer time intervals are needed to substantiate the uniformity of the loggerhead sex ratio in US coastal waters.

The St. Lucie Plant canal capture program has provided new data and augmented existing data relating to the biology and ecology of sea turtles. As the data base continues to grow, it should provide invaluable insight into seasonal abundances, sex ratios and population structures of "local" loggerhead and green turtle populations. Because of its long-term nature, the St. Lucie Plant canal capture program may constitute one of the most important gauges available for monitoring sea turtle population dynamics over time. This information will hopefully play an important role in modifying existing and developing new management plans to ensure the continued survival of sea turtles inhabiting US coastal waters.

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IMPINGEMENT CHANGES AFTER THE
INSTALLATION OF A FIXED-SCREEN FISH
DIVERSION STRUCTURE

Chris Benedict
Carolina Power & Light Company
Brunswick Biology Laboratory
P.O. Box 10429
Southport, NC 28461

ABSTRACT

Studies conducted at Carolina Power & Light Company's Brunswick Steam Electric Plant in southeastern North Carolina from 1977 through 1982 showed impingement of organisms. In an effort to reduce impingement, a fixed-screen fish diversion structure was installed in 1983. The fish diversion structure reduced total impingement over 40% by number and 70% by weight, and the dominant species impinged changed from Atlantic menhaden (Brevoortia tyrannus) to bay anchovy (Anchoa mitchilli).

INTRODUCTION

Power generating facilities use large quantities of water in the production of electricity which may result in the impingement of aquatic organisms as the intake water is filtered through traveling screens. Many investigators have reported impingement study results and the effects on the aquatic community (Uziel and Hannon, 1979). Various types of barrier and diversion methods have been utilized to exclude organisms from plant cooling systems. Behavioral barriers using electrical fields, sound, light, water jets, or air bubbles have been tested as fish diversion systems (Ray et al., 1976). These barrier systems are successful only in low-flow conditions because in each case the organism must be able to detect and actively swim away from the barrier. Various physical diversion structures--such as traveling screens, drum screens, perforated dikes, and fixed screens--filter the organisms from the intake water (Dorn and Johnson, 1981; Mussalli, 1984). The size and configuration of the openings in a diversion affect the number and the size of organisms impinged. The success of a diversion structure depends on the design, configuration, water velocity, water volume, and affected species. Each diversion structure design needs to be individually examined for the effectiveness of diverting organisms from impingement (McGroddy et al., 1985).

The Brunswick Steam Electric Plant (BSEP) is a two-unit nuclear power station located on the Cape Fear River estuary in southeastern North Carolina. The plant is situated at the end of a 4.9-km intake canal and draws water from the Cape Fear River ship channel (Figure 1). Normal intake flows for summer ($> 18.3^{\circ}\text{C}$ water temperature) and winter ($< 18.3^{\circ}\text{C}$) operation are 51.8 and 34.3 m^3/sec , respectively. Flows as high as 65 m^3/sec have been reported during some periods of prediversion sampling, while 20 m^3/sec was recorded during the lowest prediversion flow condition. These differences are due to variable plant operation.

During the late 1970s, large numbers of Atlantic menhaden (*Brevoortia tyrannus*) entered the intake canal and occasionally caused plant operation problems by obstructing the water flow through the traveling screens.

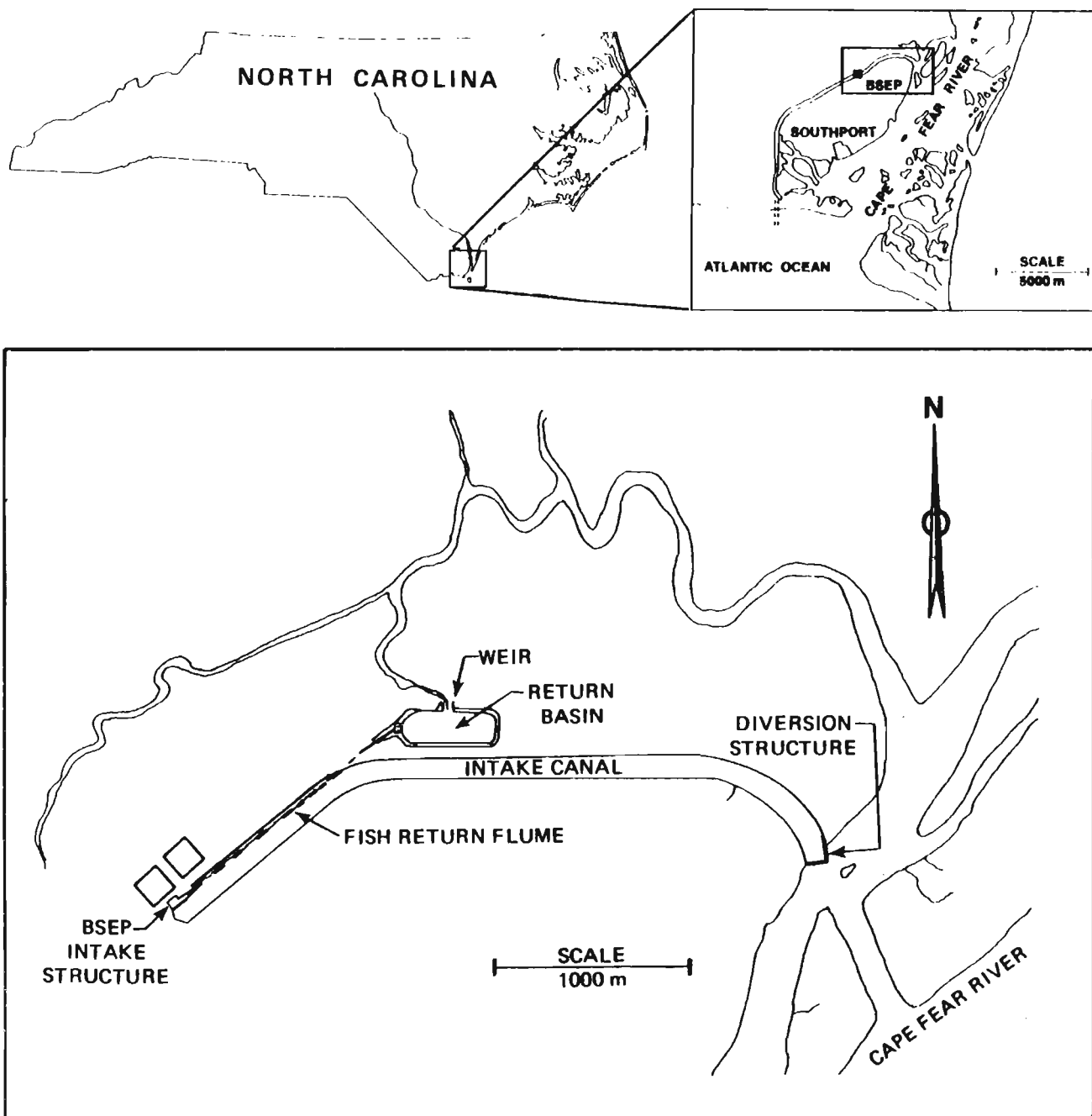


Figure 1. Location of the Brunswick Steam Electric Plant (BSEP) intake canal and diversion structure.

A test diversion structure made of framed hardware cloth and wooden pilings had been successful in reducing Atlantic menhaden catches during 1979 (Hogarth and Nichols, 1981). As a result of this study, a permanent fixed-screen fish diversion system became fully operational in January 1983. The concrete and metal diversion structure, located 3.5-km from the plant, consists of 134 screen panels 1.2 x 3.0 m in size. The diversion screens are 90% copper and 10% nickel expanded metal mesh having 9.4-mm openings. The diversion structure is "V" shaped which increases the screen surface area, decreases the through-screen velocity, and allows for some tidal flushing of debris (Figure 2).

The purpose of this paper is to describe the changes in impingement as a result of the installation of the fixed-screen fish diversion structure.

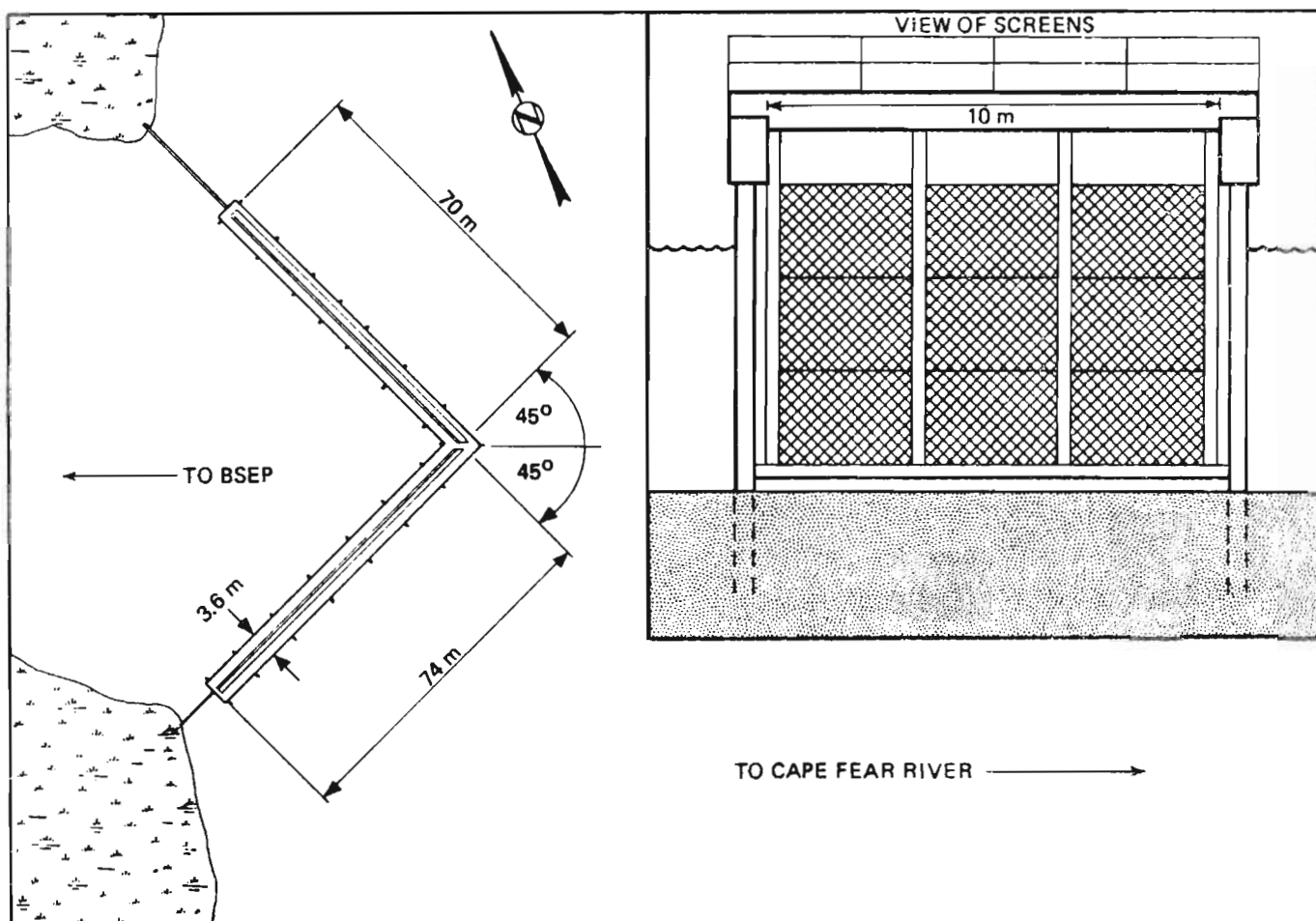


Figure 2. Overhead view of diversion structure and inset of copper-nickel screen panels.

MATERIALS AND METHODS

Impingement studies have been conducted at the BSEP since 1974. Beginning in 1977 samples were collected during full two-unit operation. A holding pen or collection basket with a 9.4-mm mesh opening was used to collect the sample. Organisms were identified to the lowest practical taxon (usually species); then length and weight were recorded. Historically, fish greater than 40-mm standard length and blue crabs with carapace widths greater than 25 mm were impinged on the 9.4-mm circulation screens. After the 1983 installation of 1-mm fine-mesh screening on four of the eight circulation screens, the cutoff lengths were imposed to separately analyze larval (< 41 mm) and juvenile/adult (> 40 mm) impingement. All impinged organisms were collected for one 24-hour period each week. Monthly and annual impingement estimates were computed from the 24-hour samples (CP&L 1985, 1986). The data are presented as number and weight of organisms per million cubic meters of plant cooling water entrained.

For reporting purposes, 1977, 1978, 1980, 1981, and 1982 will be discussed as prediversion data, while 1984 and 1985 will represent the postdiversion data. A temporary prototype diversion structure was installed in 1979 to test the feasibility of reducing impingement by diverting organisms away from the intake canal; thus 1979 data was not used. Furthermore, the 1983 data were not used because modifications to the BSEP intake structure prevented impingement sampling for half the year.

RESULTS

The installation of a fixed-screen fish diversion structure in the BSEP intake canal changed the number and weight of organisms and the rankings of species impinged. Five species represented over 80% of the total catch in six of the seven years which were examined (Table 1). In the other year (1982), gizzard shad (*Dorosoma cepedianum*) were abundant. This anomalous year was perhaps due to high freshwater river flow which transported freshwater species such as gizzard shad into the lower estuary.

Atlantic menhaden (*Brevoortia tyrannus*) was the most abundant species impinged for all prediversion years with an annual mean catch of 5481 fish per million cubic meters of water entrained. After the installation of the diversion structure, the Atlantic menhaden catch was reduced 93% in number and 91% in weight (Table 2).

Other species also showed a substantial decrease in impingement after the installation of the diversion structure. Impingement of spot (*Leiostomus xanthurus*) was reduced by 72% in number and 73.0% in weight from prediversion catches. The Atlantic croaker (*Micropogonias undulatus*) impingement was reduced 36% in number and 30% in weight over prediversion catches, and blue crab (*Callinectes* spp.) impingement was reduced 29% in number and 20% in weight (Table 2).

Table 1. A comparison of rankings of the five dominant species impinged by number at the Brunswick Steam Electric Plant before and after the installation of the fish diversion structure.

Species	Prediversion					Postdiversion	
	1977	1978	1980	1981	1982	1984	1985
<i>Brevoortia tyrannus</i>	1	1	1	1	1	2	7
<i>Anchoa mitchilli</i>	2	2	2	2	3	1	1
<i>Leiostomus xanthurus</i>	3	4	4	3	6	8	5
<i>Micropogonias undulatus</i>	5	9	5	15	5	6	3
<i>Callinectes</i> spp.	4	3	3	4	4	3	2
Total catch	85%	81%	81%	85%	66%	81%	90%

Table 2. Total number and weight (kg) of the five most abundant species impinged per million cubic meters of cooling water entrained at the BSEP during prediversion and postdiversion periods including the percent change between periods.

Species	Prediversion						Postdiversion			Change
	1977	1978	1980	1981	1982	Mean	1984	1985	Mean	
<i>Brevoortia tyrannus</i>										
Number	5,636	6,003	3,235	3,285	9,244	5,481	726	64	395	- 92.8%
Weight (kg)	39.1	41.1	34.8	31.4	47.4	38.8	5.8	0.9	3.4	- 91.3%
<i>Anchoa mitchilli</i>										
Number	2,371	1,336	1,944	2,406	883	1,788	3,356	5,166	4,261	+ 138.3%
Weight (kg)	3.6	2.5	2.5	2.6	1.1	2.5	2.2	5.1	3.7	+ 48.0%
<i>Leiostomus xanthurus</i>										
Number	533	313	233	474	255	362	99	107	103	- 71.5%
Weight (kg)	5.0	3.9	2.7	5.5	1.6	3.7	0.6	1.3	1.0	- 73.0%
<i>Micropogonias undulatus</i>										
Number	266	163	224	40	300	199	122	130	127	- 36.2%
Weight (kg)	2.5	0.7	0.6	0.5	0.8	1.0	0.5	0.8	0.7	- 30.0%
<i>Callinectes</i> spp.										
Number	271	536	342	162	443	351	318	183	251	- 28.5%
Weight (kg)	3.6	7.6	3.6	3.1	4.8	4.5	4.7	2.4	3.6	- 20.0%
Total catch										
Number	10,697	10,371	7,368	7,494	16,761	10,538	5,693	6,248	5,971	- 43.3%
Weight (kg)	64.6	72.4	51.1	50.4	64.7	60.6	18.7	15.8	17.3	- 71.5%

Bay anchovy (*Anchoa mitchilli*), which had been second in abundance during most of the prediversion years, became the dominant species in postdiversion years (Table 1). This increase in rank was not only a result of the decrease in the Atlantic menhaden catch but also a 138.3% increase in number and 48.0% increase in weight of bay anchovy over prediversion catches (Table 2).

The prediversion annual mean catch of 10,538 organisms with a weight of 60.6 kg per million cubic meters of cooling water was reduced to 5,971 organisms 17.3 kg per million cubic meters of cooling water. Total impingement was reduced 43% in number and 72% in weight in the postdiversion structure years.

DISCUSSION AND CONCLUSIONS

Impingement during the prediversion years, on occasion, threatened plant operation. Large schools of Atlantic menhaden entered the intake canal during winter months and substantially increased impingement. Other species--including bay anchovy, spot, Atlantic croaker, and blue crab--also contributed to impingement. The necessity for a fish diversion system at the BSEP was evident after several years of impingement sampling.

Fine-mesh screens (1-mm) were installed on four of the eight traveling screens at the BSEP prior to the postdiversion impingement studies. The smaller mesh size resulted in the impingement of those bay anchovies which previously could have gone through the screens. Additionally, the diversion structure did not exclude bay anchovy from the intake canal as it did larger organisms. Consequently, bay anchovy catches increased 138% during postdiversion years, while total impingement was reduced 43%. The fine-mesh screens impinged bay anchovies in postdiversion years, thus reducing the apparent effectiveness of the diversion structure on the number of organisms impingement. Total impingement was, however, reduced 72% by weight, which is a better measure of diversion structure effectiveness because the larger organisms were excluded from the impingement catch by the diversion structure.

Postdiversion impingement was reduced but not eliminated. Postlarval organisms were entrained through the diversion screens and probably utilized the intake canal as a nursery area. As the individuals grew and moved throughout the canal, they were susceptible to impingement. Occasionally the diversion screen panels would fail due to debris buildup caused by storms, lunar tides, or mechanical problems with the maintenance system. During these unusual occurrences, juvenile and adult organisms may have entered the intake canal and subsequently impinged.

The fixed-screen fish diversion structure substantially reduced Atlantic menhaden impingement. Spot, Atlantic croaker, and blue crab impingement also showed reductions in both number and weight. These substantial reductions in impingement and a reduced threat to plant operations illustrate the success of the BSEP fixed-screen fish diversion structure.

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FINE MESH SCREENS:
A NEW APPROACH TO PREVENTING ENTRAINMENT

D. Bruzek
K. Mahadevan
Mote Marine Laboratory
1600 City Island Park
Sarasota, Florida 34236

V. Brueggemeyer
E. Taft
Tampa Electric Company
P.O. Box 111
Tampa, FL 33601

ABSTRACT

In order to minimize losses of aquatic organisms at Tampa Electric Company's Big Bend Station, a novel fine-mesh screening system was installed at existing Unit 3 and new Unit 4. Six continuously traveling water screens, incorporating 0.5-mm screen mesh and specially designed organism troughs and spray washes, screen the entire flow of 1,080 cfs. Organisms washed from the screens pass via a common trough to a sump, from which they are pumped to a remote return location.

The design of the facility is based on the results of prototype screen testing at Big Bend Station in 1980 and 1981. In these studies a full-scale, prototype screen was evaluated over many months to determine the survival of selected Representative Important Species, including the early life stages of various fish and invertebrates, and to optimize the design of several key screen components. These studies indicated high survival of abundant fish eggs and invertebrates. On the basis of these results, the two-unit, fine-mesh screen design was approved by U.S. E.P.A. and was constructed.

In 1985 studies were conducted to evaluate survival of organisms recovered from the full-scale system. Results indicate that survival of fish eggs and larvae, to a great extent, exceeds estimates obtained from the prototype. Invertebrates suffer slightly higher than predicted mortality in the full-scale system, although survival is high.

This paper will present a description of the Unit 3 and 4 fine-mesh screen facility, will present results of prototype and full-scale studies, and will offer explanations for differences in observed survival rates between the prototype and full-scale installations.

INTRODUCTION

In 1985 Tampa Electric Company (TEC) placed in commercial operation a fourth generating unit (Big Bend Unit 4) at the Big Bend Station. The site is located on the eastern shore of Tampa Bay in North Ruskin, Florida. Region IV of the U.S. Environmental Protection Agency (EPA) expressed concern prior to construction of the unit over losses of organisms due to entrainment into the cooling water system. Accordingly, early in 1979 TEC conducted a prototype evaluation of a fine mesh screening system which could be utilized to protect small organisms at this site as part of the 316 Demonstration. The prototype studies were performed by Stone & Webster Engineering Corporation and Mote Marine Laboratory.

Preliminary studies in 1979 indicated that the concept of fine screening at Big Bend Station warranted further investigations in 1980. To ensure the validity of data to be obtained, it was decided that the 1980 test facility would be a full-scale, prototype traveling screen including all features of an in-service installation. Design efforts began in late summer with model studies which were conducted to optimize the screen's hydraulic characteristics and organism collection system. Construction of the prototype screen system was completed in time to initiate biological testing at the beginning of the entrainment season in March.

Biological investigations were conducted from March through August 1980. The study was conducted in three phases: Phase 1 (March) consisted of a shakedown period; Phase 2 (April 1 to May 16) involved daily sampling (5-day week) with one series of night samples per week; Phase 3 (June through August) consisted of sampling one week per month; various supplemental studies were also conducted in Phase 3.

During the study program, emphasis was placed on obtaining data for the following Representative Important Species (RIS):

<u>Common Name</u>	<u>Scientific Name</u>
Bay anchovy	<u>Anchoa mitchilli</u>
Black drum	<u>Pogonias cromis</u>
Silver perch	<u>Bairdiella chrysoura</u>
Spotted seatrout	<u>Cynoscion nebulosus</u>
Scaled sardine	<u>Harengula jaguana</u>
Tidewater silverside	<u>Menidia beryllina</u>
Stone crab	<u>Menippe mercenaria</u>
Pink shrimp	<u>Penaeus duorarum</u>
American oyster	<u>Crassostrea virginica</u>
Blue crab	<u>Callinectes sapidus</u>

Based on the positive results obtained from the prototype screen system, EPA determined that Unit 4 could be constructed with a once-through cooling system provided that a fine-mesh screening system be incorporated into the cooling water intakes of both Units 3 and 4. Unit 4 went into commercial operation in February 1985, and a biological evaluation of the full-scale system was conducted by Mote Marine Laboratory in 1985-86.

This paper presents the results for organism survivability of both the prototype and full-scale studies, discusses the similarities and differences in these results, and offers suggestions on why species-specific survival rates differed in some cases.

MATERIALS AND METHODS

Prototype Screen System Design

The prototype screen was located in the intake canal upstream of the existing Units 1 through 3 traveling screens and pumps (Figure 1). The screen was situated on a test platform connected to land by an existing bridge. Laboratory facilities were located on the north side of the intake canal. Ambient intake water was delivered to the laboratory via a series of pumps and filters located at the test platform.

Figure 2 shows a more detailed plan of the screen operating deck. The prototype screen was of the dual-flow type and incorporated all of the features required for fine screening. Seals were incorporated

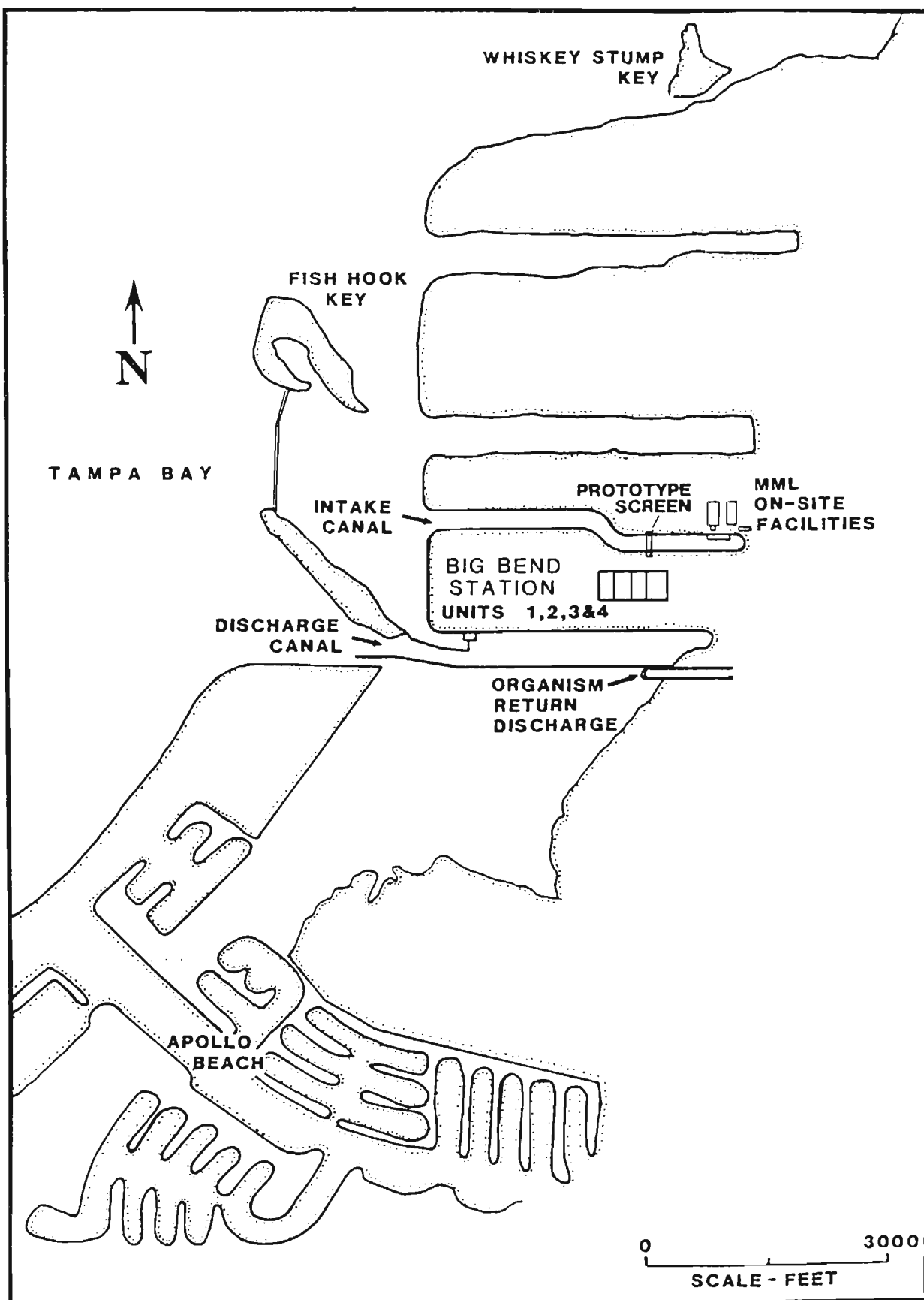


Fig. 1. Big Bend Power station site.

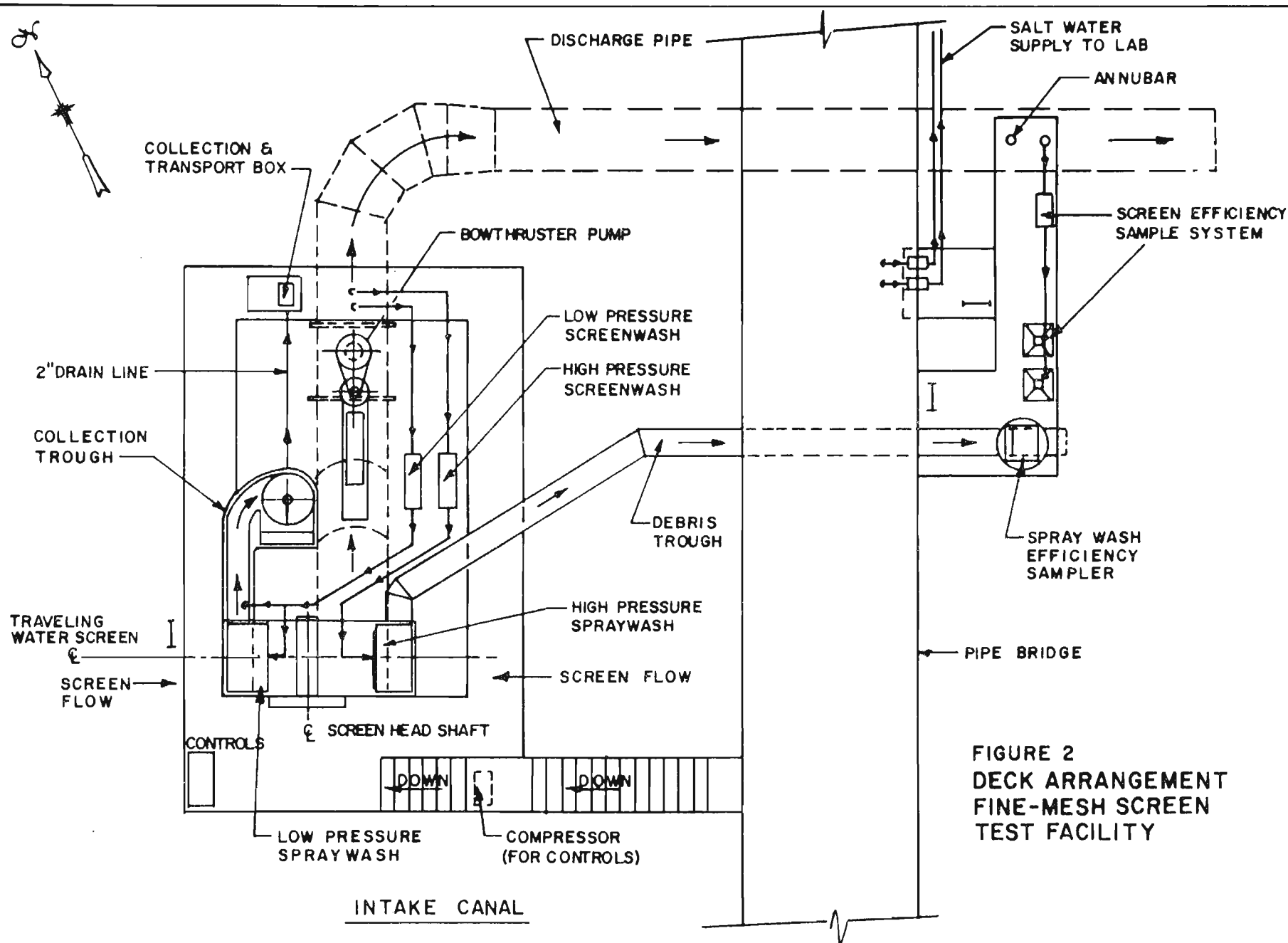


FIGURE 2
 DECK ARRANGEMENT
 FINE-MESH SCREEN
 TEST FACILITY

between screen baskets and between the baskets and the side frames to minimize the passage of organisms through these areas. The screening medium was 0.5 mm square mesh made of woven-monofilament polyester, and each of the 48 screen baskets were 0.6 m wide by 0.6 m high.

The screen was capable of operation at speeds of up to 8.5 m/min. On automatic control the screen was designed to run continuously at 2.1 m/min; as the head differential across the screen would reach 10 and then 15.2 cm, the speed would increase to 4.3 and 8.5 m/min, respectively. On manual control the screen could be operated at speeds between 0 and 8.5 m/min. Three speeds of 2.1, 4.3 and 8.5 m/min were selected for biological evaluation. These speeds corresponded to maximum impingement durations of approximately 7, 4, and 2 minutes, respectively.

Flow through the screen was supplied by an in-line pump (an adapted ship bow thruster) located under the test platform and connected directly to the screen via a transition section (Figure 3). The pump was belt-driven by a 250-hp motor located on the deck. Assorted pulleys allowed for flow adjustments. For the purpose of this study, pulleys were selected to achieve velocities of 15.2 and 30.5 cm/s.

The discharge flow from the pump, composed of fine-screened water, was conveyed by a pipe to a location far enough downstream in the intake channel to prevent recirculation (Figure 2). At the point where the pipe passed under the bridge, two taps were installed in the pipe to allow for the insertion of a flow-measuring device (for verification of pump flow rate) and an organism sampler which was used to determine the collection efficiency of the fine-mesh screen. A work platform was provided directly above this location to allow for the recording of pitometer measurements and the collection of biological samples.

The test screen incorporated shallow lifting buckets on each 0.6-m wide screen basket which retained approximately 2.54 cm of water. A low-pressure (10 psi) spray header located on the ascending side of the screen acted to remove organisms from the screen mesh surface and lifting buckets. A high-pressure (55 psi) spray header was located on the descending side of the screen to remove any remaining debris into a separate trough. A screenwash pump with a strainer was located on the operating deck and took suction from the filtered water (bow thruster

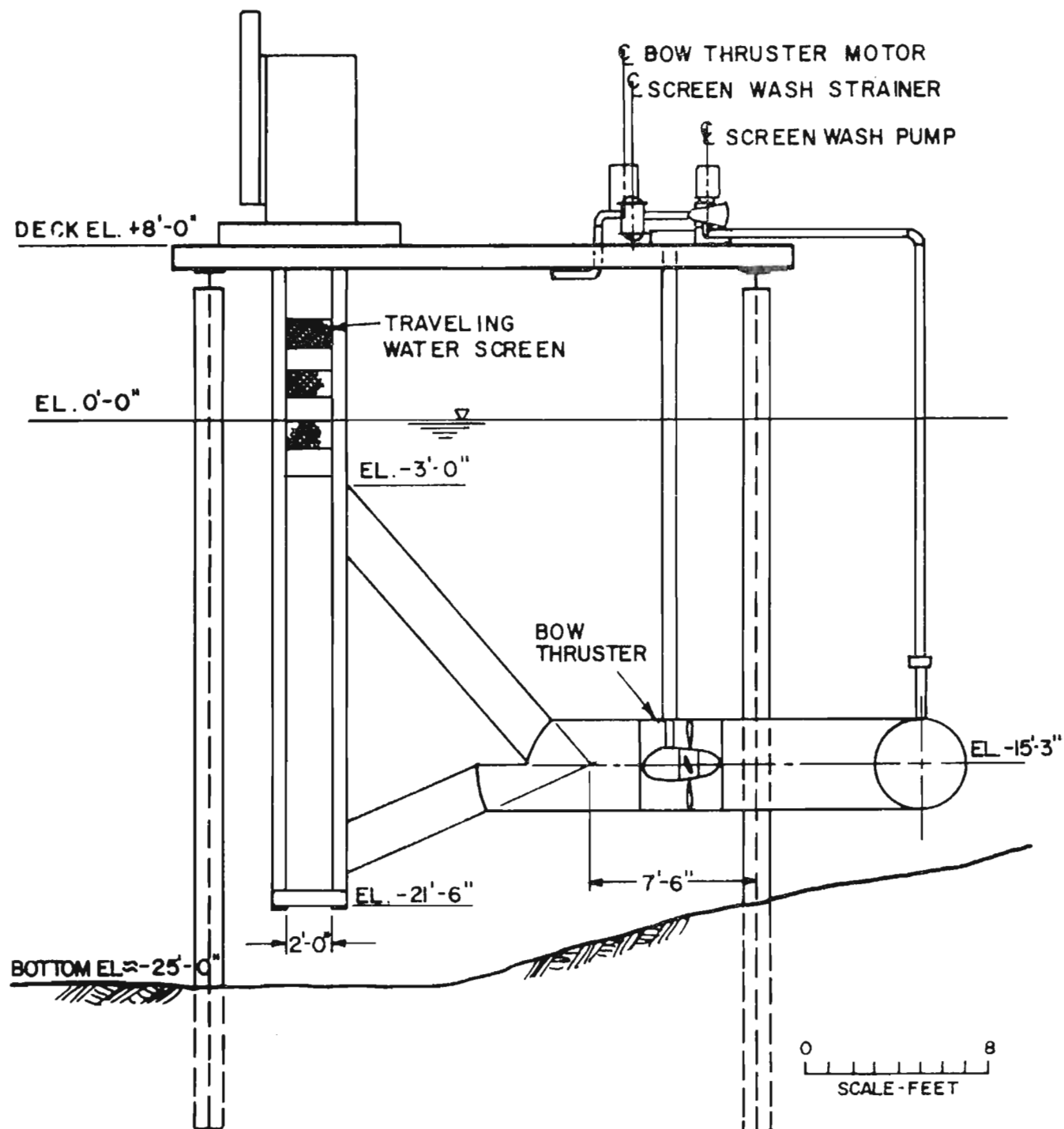


FIGURE 3
 PROFILE ARRANGEMENT
 FINE-MESH SCREEN PROTOTYPE

pump discharge). As each screen basket passed the spraywash, organisms on the mesh or in the lifting bucket were gently rinsed into a collection trough (Figure 2). Once in the trough, the organisms flowed by gravity into a primary collection tank from which they were drained into a secondary collection chamber, which also served as the container in which the organisms were transported to the laboratory.

As mentioned, a land-based laboratory was available for immediate processing of all screen samples. The laboratory facility was designed to:

1. Permit sorting, counting, identification, and determination of initial mortality of planktonic organisms.
2. Hold selected organisms for up to 96 hours to determine latent mortality.

Ambient, filtered intake water was supplied to the laboratory via a system of pumps, filters, and piping. A redundant water supply system was installed as a backup in the event of a system failure. This water served as a water bath for maintaining ambient temperatures in a system of closed holding containers.

Prototype Screen Study Methods

The primary objective of the study program was to determine the survival of organisms following impingement and removal from the prototype screen. To achieve this objective, screen samples were collected on a routine basis throughout the study program.

Essentially the same sampling procedures were utilized during all phases of the study. As discussed previously, tests were conducted at screen travel speeds of 2.1, 4.3 and 8.5 m/min and approach flow velocities of 15.2 and 30.5 cm/s. Therefore, a complete series of tests involved screenwash collection at each of the six velocity/screen speed combinations in the following matrix:

	Screen Travel Speed (m/min)		
	2.1	4.3	8.3
Approach Velocity (cm/s)			
15.2	X	X	X
30.5	X	X	X

Within a given day the timing of the sampling procedures was such that six samples could be collected and processed in the laboratory. Since the process of changing pulleys to achieve different test velocities required several hours, it was not feasible to conduct tests at both velocities within a single day. Therefore, the pulleys were changed at the end of every second day of testing at a given velocity. Within a day, two replicates of a one-velocity/three-screen speed portion of the matrix were obtained.

During Phase 1, tests were conducted five days per week (up to six tests per day) during the daytime hours (approximately between the hours of 0900 and 1500). At the beginning of Phase 2, the sampling strategy was modified to include night collection. The intent of night sampling was to determine whether species/lifestage occurrence, abundance and/or mortality differed between day and night. Under this mode, sampling was conducted Monday through Thursday during the daytime, and Thursday night between approximately 1900 and 2400 hrs. During Phase 3, the strategy continued with screen sampling being conducted one week per month (June, July, and August).

Regardless of phase or sampling strategy, the same procedures were utilized in all tests. Prior to sampling, the bow thruster pump and screen (Figure 3) were set at the desired operating point for the specific test being conducted. The low-pressure spraywash was preset at 10 psi and was then shut off until the sample was taken. Once the screen was in full operation and the movement of water and organisms through the test facility was in steady-state, sampling was initiated by turning on the low-pressure spray, allowing the contents of a predetermined number of screen baskets to be rinsed into the collection trough, and then shutting off the spray. The number of baskets washed differed for each water velocity/screen travel speed condition such that under each condition, the total volume of water sampled by the prototype screen was equivalent.

Organisms washed into the collection trough were carried into a primary collection area (Figure 2) which contained a screened overflow (0.25 mm mesh). In this area, large debris (leaves, shells, ctenophores)

could be removed as the area drained. Once the water level reached the bottom of the overflow screen, the sample was concentrated to the point where it could be drawn down into the secondary collection and transport chamber where the sample was further concentrated. The drawdown was slow and gentle, in order to avoid stress due to high velocities and turbulence in the drain line and container. Once the sample had been completely transferred, the container was covered and transported to the land-based laboratory for sorting and latent effects studies.

In addition to screenwash samples, control organisms were collected and held (as for other tests) for comparison to latent mortality values experienced among organisms collected by the screen. The control samples were taken by lowering a standard 1 meter mouth diameter, 505 micron mesh net from a walkway bridge over the intake canal. Enough line was released to allow the net to sink to mid-depth, where it was allowed to fish for fifteen minutes. The continual flow of the intake canal was sufficient enough to allow for adequate sampling from the stationary net. During Phase 1, control organisms were collected one day per week, while in Phases 2 and 3, controls were collected twice weekly.

As mentioned, all samples collected from the prototype screen and control station were transferred to the land-based laboratory where they were sorted and held. Once the transport container was delivered to the laboratory, the following procedures were carried out:

1. The transport container was placed in a water bath and temperature was recorded.
2. When RIS were abundant, the sample was gently stirred or agitated to obtain a homogeneous distributions of organisms, and a volumetric subsample was drawn off.
3. The primary sample was maintained in a water bath and held for later processing.
4. The subsample (in water bath) was sorted immediately into species/lifestages, concentrating on RIS species first.
5. Up to five individuals of each species/lifestage were placed in a separate container for transfer to the holding area; organisms were sorted to the lowest taxonomic level possible without delaying initiation of holding or adding incremental stress to the

test organisms; crab zoea and megalops were held individually in lots of 48.

6. A species/lifestage sample consisted of 30 organisms (48 crabs); therefore, once six lots of a species/lifestage were counted and recorded, they were transferred immediately to the holding area for the latent effects study.
7. Sorting continued until samples of all RIS lifestages had been placed in holding containers; the location of all organisms was carefully documented.
8. The remainder of the subsample was sorted by species, recording numbers of live, stunned, and dead organisms on data sheets. (These data were used as part of "initial survival" determinations.)
9. After sorting was completed, the primary sample was preserved for later analysis, if needed.
10. When organism densities were low, several additional subsamples were taken in order to obtain 30 live organisms of available RIS for holding. In this case, only the first subsample was completely sorted for live, stunned, and dead organisms; the remaining subsamples were drawn only to bring the total of each RIS lifestage to 30 (48 crabs), if possible. All organisms now in the holding facility were checked at 3, 6, 12, 18, 24, 36, 48, 60, 72, 84, and 96 hr; the number live and dead at each check was recorded, as well as any abnormal occurrences (e.g., missing organisms).
11. All organisms which died during the holding period were preserved in labeled vials for final identification at a later date.
12. At the completion of the holding period, the final number of live and dead organisms was recorded and each group was preserved in a labeled vial.
13. Water quality parameters were recorded frequently during all holding experiments; parameters recorded included salinity, temperature, dissolved oxygen, pH, and ammonia.
14. At the end of each latent-effects test, all vials were rechecked and all organisms were identified (to species, if possible), categorized to lifestage and counted.

These procedures were very effective in permitting an accurate accounting of most organisms from collection through final identification.

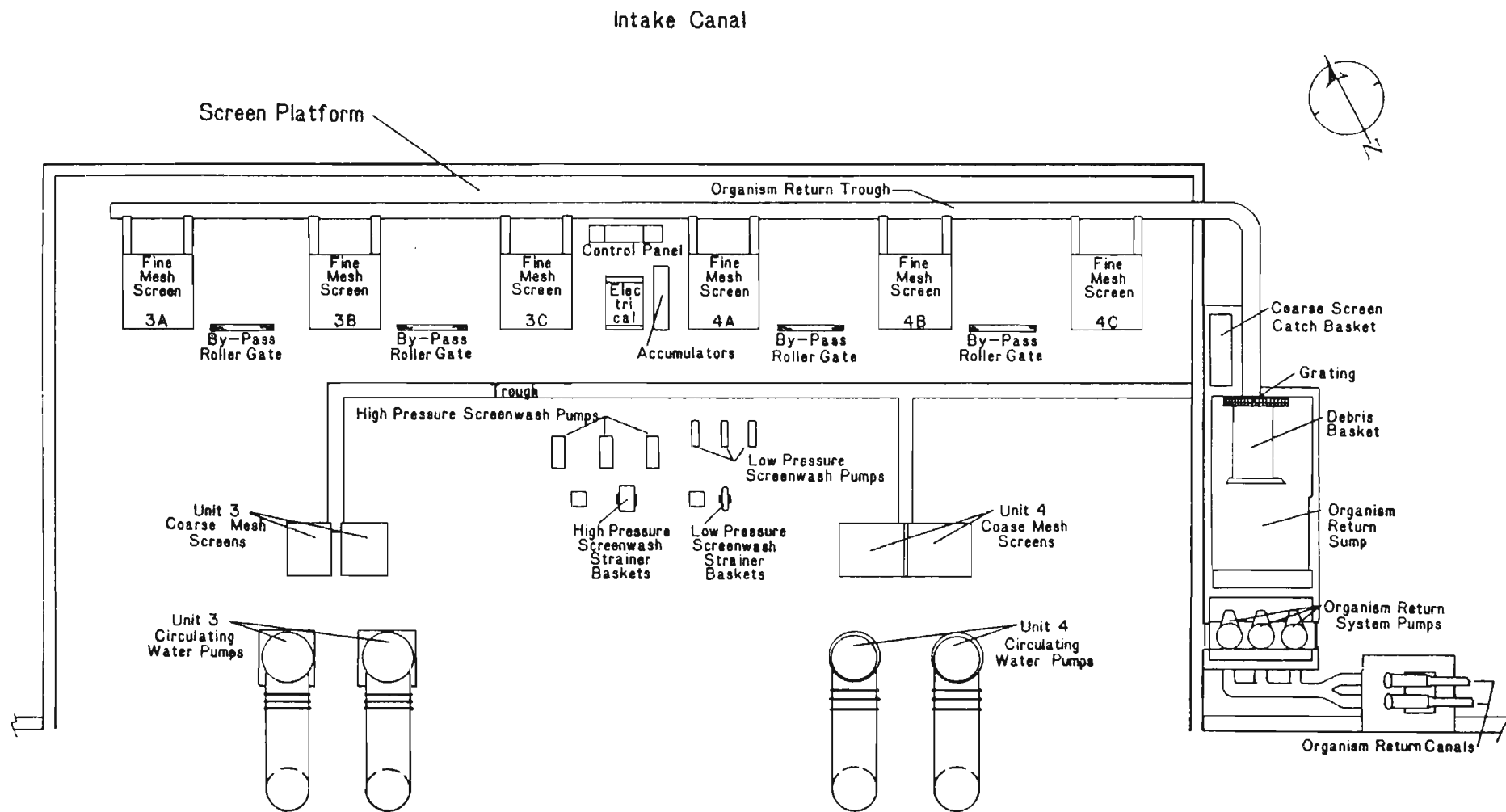


Figure 4
Full-Scale Fine Mesh Screen
Deck Arrangement

Full-Scale System Design

The intake design was developed on the basis of applying fine-mesh screens to existing Unit 3 and proposed Unit 4. Sheet piling encloses the pump screenwell area, creating a forebay between the fine-mesh screens and the pumps. A layout of the new screening structure is shown in Figure 4.

Each of the six (three for each unit) dual-flow fine-mesh screens were fitted in the sheetpile wall, and the intake flow passes through the screens and discharges into the forebay. The discharge opening was sized to achieve a uniform flow distribution through the two parallel screen faces. To ensure proper hydraulic conditions at the four circulating water pumps and a relatively uniform flow through each of the six screens, the forebay extends from the seawall about 18.3 m into the intake channel. The pumps were positioned to aid proper flow distribution so that each set of two circulating pumps draws from three screens.

Stop gates close the discharge opening in the event that a screen must be stopped or removed for maintenance. In case of emergency conditions, roller gates were installed in the sheetpile wall between screens and open to supply water to the pumps if the pressure head differential across the screens exceeds 0.46 m. To screen the flow and protect the pumps and condenser under this emergency operating mode, the existing coarse-mesh 9.5 mm screens were maintained on Unit 3, and new similar coarse-mesh screens were installed on Unit 4.

The dual-flow screen design with 9.5 mm mesh has been used successfully at Big Bend Units 1 through 3 for many years and, based on testing to date, has the necessary features for protecting small organisms when fitted with fine screens. The number and size of the screens were determined on the basis of achieving a screen face velocity (approach velocity) of 15.2 cm/s at water level El 0.0 MLW. Based on the 15.2 cm/s velocity criterion, six 3.05 m wide screens were installed. Each screen is submerged to El -8.5 m. Each of the 53 screen baskets is equipped with easily replaceable 0.5 mm mesh panels and fish collection buckets. The fine-mesh requires continuous screen operation to prevent clogging and to limit the time that organisms are impinged.

Each fine-mesh screen is capable of operating continuously at speeds of up to 8.5 m/min. On automated control the screen runs continuously at variable speed. As the head differential across the screen reaches 10 cm and then 15 cm, the speed automatically increases to 4.3 and 8.5 m/min, respectively.

In addition, each screen is equipped with both a low pressure (approximately 5-10 psi) organism spray wash header on the ascending side and a high pressure (40 to 60 psi) spray wash header for debris removal on the descending side of the screen. Separate pumps supply water to each header and take suction from the fine-screened water downstream of the circulating water pumps. These nozzles have 12.7 mm diameter orifices and are evenly spaced at 0.3 m intervals over the 3.05 m header length to ensure uniform spray pattern across the screen baskets.

Although the low and high pressure sprays wash organisms and debris into separate troughs, all troughs manifold into a common sluiceway that carries organisms and debris to a release point in the northern Apollo Beach embayment (Figure 1). This maximizes the opportunity for survival of any organisms not removed by the low pressure spray but subsequently removed by the debris spray.

The organism return trough was sized to achieve a transport velocity in the range of 37 to 255 m/min at a water depth of from 15 to 30 cm. Total length of the return flume is 630 m.

The Unit 3 operating deck from which the dual-flow fine-mesh screens hang was expanded to accommodate the six screens. As with Units 1 and 2 structure, the expanded operating deck is supported on piles at El +3.54 m. The channel bottom would be maintained in the screen structure area at the existing depth of El -9.65 m.

The cost required to add a fine-mesh screen system to Units 3 and 4 was \$9.9 million.

Full-Scale System Study Methods

Methods used during this study were essentially similar to those used during the Fine Mesh Screen Prototype study conducted in 1980.

An on-site facility was established at the Big Bend station for the fine mesh screen survivability studies. This facility consisted of a

test trailer with a flow-through seawater system, an adjacent office trailer, and a sample wash and storage trailer. The on-site facility was located along the intake canal near the fine mesh screens. All live meroplankton sorting and holding of organisms for 48 hours was conducted at this facility.

The test facility trailer was the same one used for the prototype study and was equipped with a flow-through seawater system consisting of two 1/2 hp pumps that pump intake canal water through four sand filters up to a 1130 l head tank. The filtered seawater would then gravity feed from the head tank through the holding facility.

The survival studies were designed to concentrate on the same target species which were found in high abundance in the 1980 studies. In addition, eggs and larvae of other fish species which appeared in large numbers along with any juveniles that were collected were held in latent mortality experiments. Because of the difficulty of distinguishing larvae of Menippe and Upogebia during live sorting, zoea and larvae of a number of crustacean species were held in latent mortality tests.

These studies commenced the first week of March 1985. Field studies were conducted each week from March 5, 1985 until October 1, 1985, for a total of 31 weeks of sampling. Two collections per day and two per night were taken one day per week at the screenwash and organism return discharge stations. One collection per day and one per night were taken at the control station one day each week.

The low and high pressure screenwash of the fine mesh screens empties into a common trough which flows to a sump at the beginning of the organism return canal. The screenwash station was sampled by suspending a 1 meter mouth diameter, 505 micron mesh net in front of the trough where it empties into the sump basin. A bucket attached to the cod end of the net held it open, reducing damage to the organisms as a result of friction against the net. The net was fished for three minutes, and all organisms retained were carefully rinsed down into the bucket. The sample was then immediately returned to the test trailer for sorting.

The organism return discharge (ORD) pipes exit at the end of a walkway which extends in the Apollo Beach northern embayment. The ORD station was sampled using a specially constructed 505 micron mesh net supported by a floating rectangular frame (1 m x 1.6 m). The net was positioned under the flow from the discharge pipe, and organisms retained were carefully rinsed down to a bucket attached to the cod end of the net. The net was fished for three minutes, and the sample was then immediately returned to the test trailer for sorting.

The control sample was taken by lowering a standard 1 meter mouth diameter, 505 micron mesh net from a walkway bridge over the intake canal. Enough line was released to allow the net to sink to mid-depth, where it was allowed to fish for 15 minutes. The continual flow of the intake canal was sufficient enough to allow for adequate sampling from the stationary net. Organisms retained by the net were carefully rinsed to the cod end of the net, and the contents were then carefully poured into a bucket of ambient intake water. The sample was then immediately returned to the test trailer for sorting.

After the samples were returned to the test trailer, the procedures used to assess initial survivability and latent survivorship were identical to those outlined in the prototype study methods. The only differences were that stunned organisms were not considered in the full-scale study, and holding for latent survivorship studies was done for only 48 hours, with checks done at 6, 12, 24, 36 and 48 hours. Identifications were made at Mote Marine Laboratory through the use of standard literature sources and MML's reference collection. A voucher reference collection of taxonomically confirmed species was maintained.

RESULTS

The results of the prototype screen study are given in Tables 1 through 4 as the percent initial and latent survival of the fish and invertebrate lifestages which were collected from the prototype screen over the duration of the study. In order to conduct the data analysis, it was necessary to combine identified taxa into groups since, in most cases, a single species, genus, or even family did not occur frequently

TABLE 1

FISH EGGS
PERCENT SURVIVAL AND HATCHABILITY

	Initial Survival		Hatchability		48-HR Survival		96-HR Survival	
	Test	Control	Test	Control	Test	Control	Test	Control
Sciaenidae	75.3	98.4	94.8	99.0	84.3	91.3	69.7	82.7
<u>Bairdiella</u> <u>chrysura</u>	100	100	100	100	99.1	99.4	97.9	97.8
<u>Cynoscion</u> spp.	100	100	100	100	99.4	99.3	89.4	96.9
<u>Menticirrhus</u> spp.	100	100	100	100	99.7	100	88.4	91.5
<u>Pogonias cromis</u>	100	-	100	-	82.2	-	85.3	-
Clupeiformes	43.2	85.5	81.0	89.3	84.4	90.3	62.4	68.6
<u>Harengula jaguana</u>	45.8	99.6	92.9	98.5	82.8	92.2	45.9	27.6
<u>Anchoa mitchilli</u>	43.3	85.0	80.0	88.6	83.9	90.0	63.7	72.0

Note: Dashes indicate no observations

TABLE 2

FISH LARVAE
PERCENT INITIAL AND LATENT SURVIVAL

	Initial Survival		48-HR Survival		96-HR Survival	
	<u>Test</u>	<u>Control</u>	<u>Test</u>	<u>Control</u>	<u>Test</u>	<u>Control</u>
Sciaenidae	18.6 (108)	44.4 (6)	10.9 (26)	0 (1)	10.1(26)	0 (1)
<u>Bairdiella</u> <u>chrysura</u>	19.2 (39)	50.0 (2)	-	-	-	-
<u>Cynoscion</u> spp.	15.7 (51)	0 (1)	100 (3)	-	100 (3)	-
<u>Menticirrhus</u> spp.	0 (15)	25.0 (4)	-	-	-	-
<u>Pogonias cromis</u>	42.9 (7)	100 (1)	-	-	-	-
Clupeiformes	1.5 (278)	10.4 (11)	36.4 (11)	0(1)	36.4(11)	0 (1)
<u>Harengula jaguana</u>	0 (15)	-	-	-	-	-
<u>Anchoa mitchilli</u>	1.5 (274)	11.4 (10)	22.2 (9)	0(1)	22.2(9)	0 (1)

Notes: Number of observations is given in parentheses
Dashes indicate no observations

TABLE 3

DECAPOD ZOEAE
PERCENT INITIAL AND LATENT SURVIVAL

	Initial Survival		48-HR Survival		96-HR Survival	
	<u>Test</u>	<u>Control</u>	<u>Test</u>	<u>Control</u>	<u>Test</u>	<u>Control</u>
Caridea	94.3	76.7	85.0	86.8	50.0	43.8
<u>Upogebia affinis</u>	91.3	75.6	84.1	76.2	42.8	45.4
Brachyura	95.5	65.0	83.9	55.6	45.9	27.8
Grapsizoea	100	100	95.1	97.9	80.2	92.9
Pinnotheridae	100	100	92.2	93.4	73.0	72.1
Xanthidae	99.1	-	95.9	95.6	74.9	73.4
<u>Menippe mercenaria</u>	97.9	97.3	91.5	94.9	58.3	61.0
Paguridae	94.7	100	96.6	100	79.2	33.3

Note: Dashes indicate no observations

TABLE 4

DECAPOD MEGALOPS
PERCENT INITIAL AND LATENT SURVIVAL

	Initial Survival		48-HR Survival		96-HR Survival	
	<u>Test</u>	<u>Control</u>	<u>Test</u>	<u>Control</u>	<u>Test</u>	<u>Control</u>
Caridea	100	-	100	-	100	-
<u>Upogebia affinis</u>	100	100	97.7	100	74.3	100
Brachyura	65.1	26.7	71.8	-	15.0	-
Grapsizoea	100	100	98.1	100	93.1	91.2
Pinnotheridae	100	-	100	-	92.9	-
Xanthidae	100	100	98.3	100	94.2	96.9
<u>Menippe mercenaria</u>	100	-	100	-	100	-
Paguridae	100	-	90.0	-	80.0	-

Note: Dashes indicate no observations

enough to permit a meaningful analysis. The taxonomic level of each group was determined as the lowest level which would allow inclusion of all important taxa.

The test data indicate that the invertebrates had the highest survival, most often in excess of 90 percent; fragile fish larvae, such as the bay anchovy (Anchoa mitchilli), had low survivorship, as anticipated for larvae. However, it should be noted that survival among control larvae was also low. Further, as shown on Table 2, the small number of observations for some taxa limits the conclusions which might be drawn from these data.

The latent survivorship of those organisms which were initially alive following collection was studied in the laboratory holding experiments. The proportion of organisms which were alive at 48 hr and 96 hr was used as an index of latent survival. Both measures provide an indication of the success of the device; however, past experience suggested that high mortality at 96 hr might limit the utility of this measure.

Taxa/lifestages which were used as a basis for determining 48 and 96 hr survival are the same as those for which initial survival was determined. The mean initial survivabilities are shown in Tables 1 through 4. The results indicate that survivorship is taxa- and lifestage-specific, as is the relationship between 48 hr and 96 hr values.

The hatchability of eggs observed during the holding experiments is an index of the viability of these eggs. Since the development time of eggs for many of the species tested is quite rapid, nearly all live eggs hatched prior to 96 hr. The proportion of eggs which hatched during the holding period and the survivorship of these hatched eggs is presented in Table 1.

The results of holding experiments with control samples which had not experienced the fine-mesh screen system are also presented in Tables 1 through 4. These survivorship values indicate that natural mortality, not associated with stresses resulting from the collecting and holding procedures, is a contributing factor to the mortalities observed among organisms collected by the fine-mesh screen. Therefore, observed test

mortalities should be considered as the cumulative effects of test and natural mortality.

Analyses of variance were conducted to determine whether screen travel speed (2.1, 4.3 and 8.5 m/min), velocity approaching the screen (15.2 and 30.5 cm/s), or water temperature influenced initial or latent mortality. These analyses were limited to the most abundant taxa/lifestages: among fish, Anchoa mitchilli, Sciaenidae and Cynoscion spp. eggs and larvae were chosen; for invertebrates, Brachyura and Xanthidae zoeal and megalops stages and Menippe mercenaria zoeal stages were selected.

In general, the analyses did not explain a high proportion of the variation in mortality. The independent variables which were most often significant ($p \leq 0.05$) were water temperature and approach velocity. The temperature variable indicated decreasing survival with increasing temperature for all lifestages except fish eggs. Lowest 96 hr survival occurred during Phase 3 when the highest temperatures occurred. Temperature did not have as great an effect on 48 hr survival. It should be noted that 96 hr survival of control organisms also decreased at higher temperatures, reflecting the difficulty of maintaining these organisms for long periods under laboratory conditions.

Relative to approach velocity, the analyses indicated that velocity had a significant effect on egg hatchability and survival of Sciaenidae and on 48 hr survival of Xanthidae zoea. However, differences in survival between 15.2 and 30.5 cm/s were relatively small:

<u>Dependent Variable</u>		<u>Mean Value (%)</u>	
		<u>15.2 cm/s</u>	<u>30.5 cm/s</u>
Sciaenidae	Initial egg survival	80.8	71.4
	Egg hatchability	96.6	93.3
	48 hr survival of hatched eggs	89.5	79.3
	96 hr survival of hatched eggs	76.8	64.0
	48 hr survival of Xanthidae zoea	97.0	94.7

It must be emphasized that both approach velocity and water temperature, while statistically significant, explained a small amount of

the variability in the dependent variable. Therefore, within the range of independent variables tested, the mean survivorship values presented in Tables 1 through 4 are good indicators of the performance of the fine-mesh screen facility.

The results of the full-scale screen study are given in Tables 5 through 11 as the percent initial and latent survival of fish and invertebrate lifestages which were collected from the three stations (Screenwash, ORD, and Control) over the duration of the study. In most cases, identified taxa were combined into groups for a more meaningful analysis. The taxonomic level of each group was determined as the lowest level which would allow inclusion of all important taxa.

The test data for initial survival are presented in Tables 5 through 7. Results indicate that the invertebrates had the highest survival, with mean values in excess of 80% at all three stations. Fragile fish larvae had the lowest survival, though there was considerable variability between species and stations.

Fish egg survival was lowest at the ORD station and highest at the control station.

The test data for latent survival are presented in Tables 8 through 10. The proportion of organisms alive at 24 and 48 hours was used as an index of latent survival. Latent survival was generally high for the invertebrates, with survival highest at the control station and lowest at the ORD station. Latent survival of the fish larvae was generally around 50%, with no significant difference between stations.

The test data for hatchability are presented in Table 11. The hatchability of eggs during holding is a measure of the viability of these eggs. Nearly all viable eggs hatched before 48 hrs after collection. The hatchability of eggs was high at all three stations.

The initial and latent survival of organisms from the control stations indicate that natural mortality, not associated with stresses resulting from the fine mesh screen process, is a contributing factor to the mortalities observed among organisms collected by the fine mesh screens (Table 12). This factor must be kept in mind when considering the test mortalities obtained.

Table 5. Summary data of initial survivability tests at Screenwash Station, Big Bend Power Station (1985). (Survivability: 1.00 = 100%)

	INITIAL SURVIVABILITY		
	Total # Organisms Tested	Mean	S.D.
<u>FISH</u>			
EGGS:			
CLUPEIDAE	219	0.78	0.04
ENGRAULIDAE	21,899	0.48	0.36
SCIAENIDAE	51,202	0.63	0.03
SOLEIDAE	29	0.88	0.18
ALL EGGS:	73,349	0.69	0.18
LARVAE:			
CLUPEIDAE	2	0.50	0.71
ENGRAULIDAE	1,651	0.16	0.04
ATHERINIDAE	1	0.00	0.00
SPARIDAE	1	0.00	0.00
<u>Lagodon rhomboides</u>	--	--	--
SCIAENIDAE	1,632	0.61	0.33
<u>Bairdiella chrysoura</u>	2	0.00	0.00
<u>Pogonias cromis</u>	--	--	--
<u>Cynoscion nebulosus</u>	1	0.00	0.00
<u>Cynoscion arenarius</u>	25	0.00	0.00
<u>Leiostomus xanthurus</u>	--	--	--
BLENNIIDAE	3	0.25	0.35
GOBIIDAE	107	0.10	0.00
SOLEIDAE	35	0.00	0.00
ALL LARVAE:	3,460	0.15	0.22
<u>INVERTEBRATES</u>			
ZOEAE:			
PENAEIDAE	6	1.00	0.00
CARIDEA	1,057	0.72	0.01
PORTUNIDAE	45	0.94	0.07
PAGURIDAE	81	0.96	0.04
XANTHIIDAE	11,143	0.93	0.04
PINNOTHERIDAE	33,287	0.89	0.06
GRAPSIDAE	512	0.92	0.07
ALL ZOEAE:	46,131	0.91	0.09
MEGALOPS:			
XANTHIIDAE	25	0.77	0.33
GRAPSIDAE	149	0.82	0.20
ALL MEGALOPS:	174	0.80	0.04

Table 6. Summary data of initial survivability tests at the Organism Return Discharge Station, Big Bend Power Station (1985). (Survivability: 1.00 = 100%)

	INITIAL SURVIVABILITY		
	Total # Organisms Tested	Mean	S.D.
<u>FISH</u>			
EGGS:			
CLUPEIDAE	9	0.49	0.13
ENGRAULIDAE	3,522	0.29	0.21
SCIAENIDAE	16,693	0.40	0.04
SOLEIDAE	23	0.56	0.20
ALL EGGS:	20,247	0.44	0.12
LARVAE:			
CLUPEIDAE	2	0.00	0.00
ENGRAULIDAE	271	0.58	0.60
ATHERINIDAE	--	--	--
SPARIDAE	--	--	--
<u>Lagodon rhomboides</u>	--	--	--
SCIAENIDAE	284	0.56	0.16
<u>Bairdiella chrysoura</u>	--	--	--
<u>Pogonias cromis</u>	--	--	--
<u>Cynoscion nebulosus</u>	--	--	--
<u>Cynoscion arenarius</u>	11	0.00	0.00
<u>Leiostomus xanthurus</u>	--	--	--
BLENNIIDAE	1	1.00	0.00
GOBIIDAE	5	0.00	0.00
SOLEIDAE	1	0.00	0.00
ALL LARVAE:	575	0.31	0.41
<u>INVERTEBRATES</u>			
ZOEAE:			
PENAEIDAE	2	0.50	0.00
CARIDEA	458	0.70	0.02
PORTUNIDAE	7	1.00	0.00
PAGURIDAE	21	0.79	0.23
XANTHIIDAE	7,576	0.90	0.01
PINNOTHERIDAE	29,845	0.83	0.07
GRAPSIDAE	442	0.96	0.06
ALL ZOEAE:	38,351	0.81	0.17
MEGALOPS:			
XANTHIIDAE	5	0.50	0.71
GRAPSIDAE	109	0.76	0.08
ALL MEGALOPS:	114	0.63	0.63

Table 7. Summary data of initial survivability tests at the Control Station, Big Bend Power Station (1985). (Survivability: 1.00 = 100%)

	INITIAL SURVIVABILITY		
	Total # Organisms Tested	Mean	S.D.
<u>FISH</u>			
EGGS:			
CLUPEIDAE	60	0.73	0.38
ENGRAULIDAE	16,378	0.72	0.31
SCIAENIDAE	5,872	0.72	0.05
SOLEIDAE	3	1.00	0.00
ALL EGGS:	22,313	0.79	0.14
LARVAE:			
CLUPEIDAE	--	--	--
ENGRAULIDAE	303	0.16	0.09
ATHERINIDAE	3	0.00	0.00
SPARIDAE	--	--	--
<u>Lagodon rhomboides</u>	--	--	--
SCIAENIDAE	95	0.85	0.22
<u>Bairdiella chrysoura</u>	--	--	--
<u>Pogonias cromis</u>	--	--	--
<u>Cynoscion nebulosus</u>	--	--	--
<u>Cynoscion arenarius</u>	--	--	--
<u>Leiostomus xanthurus</u>	--	--	--
BLENNIIDAE	12	0.60	0.57
GOBIIDAE	14	0.50	0.71
SOLEIDAE	35	0.00	0.00
ALL LARVAE:	462	0.35	0.35
<u>INVERTEBRATES</u>			
ZOEAE:			
PENAEIDAE	--	--	--
CARIDEA	347	0.65	0.09
PORTUNIDAE	14	1.00	0.00
PAGURIDAE	5	1.00	0.00
XANTHIIDAE	671	0.88	0.10
PINNOTHERIDAE	2,987	0.77	0.13
GRAPSIDAE	37	0.81	0.11
ALL ZOEAE:	4,061	0.85	0.14
MEGALOPS:			
XANTHIIDAE	1	1.00	0.00
GRAPSIDAE	24	0.84	0.08
ALL MEGALOPS:	25	0.92	0.11

Table 8. Summary data of latent survivability tests for 24 and 48 hours, Screenwash Station, Big Bend Power Station (1985). (Survivability: 1.00 = 100%)

	Total # Organisms Tested	LATENT SURVIVABILITY			
		24 hrs		48 hrs	
		Mean	S.D.	Mean	S.D.
<u>FISH</u>					
EGGS:					
CLUPEIDAE	4	0.25	0.35	--	--
ENGRAULIDAE	322	0.45	0.13	--	--
SCIAENIDAE	549	0.44	0.04	--	--
SOLEIDAE	8	0.30	0.18	--	--
ALL EGGS:	883	0.36	0.10	--	--
LARVAE:					
CLUPEIDAE	28	0.59	0.12	0.44	0.16
ENGRAULIDAE	2,200	0.77	0.10	0.68	0.04
ATHERINIDAE	1	0.00	0.00	0.00	0.00
SPARIDAE	--	--	--	--	--
<u>Lagodon rhomboides</u>	1	0.00	0.00	0.00	0.00
SCIAENIDAE	1,852	0.73	0.02	0.63	0.04
<u>Bairdiella chrysoura</u>	6	0.50	0.50	0.50	0.50
<u>Pogonias cromis</u>	7	1.00	0.00	1.00	0.00
<u>Cynoscion nebulosus</u>	2	0.50	0.71	0.50	0.71
<u>Cynoscion arenarius</u>	143	0.57	0.04	0.44	0.13
<u>Leiostomus xanthurus</u>	--	--	--	--	--
BLENNIIDAE	15	0.62	0.28	0.62	0.28
GOBIIDAE	39	0.39	0.55	0.36	0.50
SOLEIDAE	941	0.98	0.03	0.96	0.04
ALL LARVAE:	5,235	0.55	0.32	0.51	0.31
<u>INVERTEBRATES</u>					
ZOEAE:					
PENAEIDAE	52	0.95	0.07	0.95	0.07
CARIDEA	1,992	0.78	0.01	0.67	0.06
PORTUNIDAE	34	0.96	0.06	0.81	0.03
PAGURIDAE	97	0.77	0.07	0.76	0.09
XANTHIIDAE	1,771	0.88	0.02	0.80	0.06
PINNOTHERIDAE	3,499	0.85	0.00	0.71	0.04
GRAPSIDAE	176	0.91	0.07	0.85	0.03
ALL ZOEAE:	7,621	0.87	0.08	0.79	0.09
MEGALOPS:					
XANTHIIDAE	215	0.99	0.02	0.97	0.04
GRAPSIDAE	1,230	0.98	0.01	0.95	0.00
ALL MEGALOPS:	1,445	0.99	0.01	0.96	0.01
JUVENILES:					
<u>Loliguncula brevis</u>	44	0.03	0.01	0.02	0.03

Table 9. Summary data of latent survivability tests for 24 and 48 hours, Organism Return Discharge Station, Big Bend Power Station (1985). (Survivability: 1.00 = 100%)

	Total # Organisms Tested	LATENT SURVIVABILITY			
		24 hrs		48 hrs	
		Mean	S.D.	Mean	S.D.
<u>FISH</u>					
EGGS:					
CLUPEIDAE	2	0.50	0.00	--	--
ENGRAULIDAE	128	0.44	0.06	--	--
SCIAENIDAE	505	0.42	0.01	--	--
SOLEIDAE	7	0.65	0.02	--	--
ALL EGGS:	642	0.50	0.10	--	--
LARVAE:					
CLUPEIDAE	3	1.00	0.00	1.00	0.00
ENGRAULIDAE	1,249	0.86	0.05	0.65	0.11
ATHERINIDAE	4	0.16	0.23	0.16	0.23
SPARIDAE	--	--	--	--	--
<u>Lagodon rhomboides</u>	2	1.00	0.00	1.00	0.00
SCIAENIDAE	1,376	0.83	0.06	0.66	0.00
<u>Bairdiella chrysoura</u>	--	--	--	--	--
<u>Pogonias cromis</u>	1	1.00	0.00	1.00	0.00
<u>Cynoscion nebulosus</u>	--	--	--	--	--
<u>Cynoscion arenarius</u>	12	0.34	0.47	0.34	0.47
<u>Leiostomus xanthurus</u>	--	--	--	--	--
BLENNIIDAE	2	0.50	0.71	0.50	0.71
GOBIIDAE	7	0.67	0.47	0.67	0.47
SOLEIDAE	873	0.93	0.09	0.90	0.11
ALL LARVAE:	3,529	0.73	0.30	0.69	0.29
<u>INVERTEBRATES</u>					
ZOEAE:					
PENAEIDAE	27	0.99	0.00	0.97	0.03
CARIDEA	2,118	0.79	0.00	0.66	0.00
PORTUNIDAE	6	0.88	0.17	0.90	0.14
PAGURIDAE	76	0.81	0.09	0.68	0.14
XANTHIIDAE	1,799	0.81	0.04	0.71	0.06
PINNOTHERIDAE	3,674	0.80	0.00	0.65	0.00
GRAPSIDAE	192	0.94	0.03	0.89	0.03
ALL ZOEAE:	7,892	0.86	0.08	0.78	0.13
MEGALOPS:					
XANTHIIDAE	72	0.93	0.07	0.92	0.08
GRAPSIDAE	1,469	0.96	0.03	0.94	0.04
ALL MEGALOPS:	1,541	0.95	0.02	0.93	0.01
JUVENILES:					
<u>Lolliguncula brevis</u>	4	0.00	0.00	0.00	0.00

Table 10. Summary data of latent survivability tests for 24 and 48 hours, Control Station, Big Bend Power Station (1985). (Survivability: 1.00 = 100%)

	Total # Organisms Tested	LATENT SURVIVABILITY			
		24 hrs		48 hrs	
		Mean	S.D.	Mean	S.D.
<u>FISH</u>					
EGGS:					
CLUPEIDAE	1	0.50	0.00	--	--
ENGRAULIDAE	30	0.51	0.21	--	--
SCIAENIDAE	109	0.56	0.06	--	--
SOLEIDAE	1	0.50	0.00	--	--
ALL EGGS:	141	0.52	0.03	--	--
LARVAE:					
CLUPEIDAE	23	0.88	0.18	0.74	0.01
ENGRAULIDAE	1,361	0.70	0.21	0.59	0.23
ATHERINIDAE	49	0.52	0.17	0.46	0.08
SPARIDAE	--	--	--	--	--
<u>Lagodon rhomboides</u>	1	0.00	0.00	0.00	0.00
SCIAENIDAE	902	0.77	0.11	0.61	0.00
<u>Bairdiella chrysoura</u>	6	0.50	0.50	0.50	0.50
<u>Pogonias cromis</u>	--	--	--	--	--
<u>Cynoscion nebulosus</u>	--	--	--	--	--
<u>Cynoscion arenarius</u>	3	1.00	0.00	1.00	0.00
<u>Leiostomus xanthurus</u>	--	--	--	--	--
BLENNIIDAE	147	0.78	0.08	0.65	0.00
GOBIIDAE	156	0.78	0.31	0.75	0.21
SOLEIDAE	285	0.95	0.01	--	--
ALL LARVAE:	2,933	0.69	0.29	0.59	0.27
<u>INVERTEBRATES</u>					
ZOEAE:					
PENAEIDAE	1	1.00	0.00	1.00	0.00
CARIDEAN	1,362	0.95	0.03	0.88	0.08
PORTUNIDAE	14	1.00	0.00	0.67	0.47
PAGURIDAE	44	1.00	0.00	0.82	0.10
XANTHIIDAE	750	0.90	0.05	0.85	0.06
PINNOTHERIDAE	1,281	0.86	0.06	0.74	0.13
GRAPSIDAE	107	0.93	0.03	0.88	0.00
ALL ZOEAE:	3,559	0.95	0.06	0.83	0.11
MEGALOPS:					
XANTHIIDAE	28	1.00	0.00	1.00	0.00
GRAPSIDAE	261	1.00	0.00	1.00	0.00
ALL MEGALOPS:	289	1.00	0.00	1.00	0.00
JUVENILES:					
<u>Lolliquncula brevis</u>	1	0.00	0.00	0.00	0.00

Table 11. Summary data for hatchability tests of fish eggs from the full-scale study, Big Bend Power Station (1985). (Hatchability: 1.00 = 100%)

SCREENWASH STATION

	(n)	DAY	(n)	NIGHT
Clupeidae	6	0.83	5	0.71
Engraulidae	1299	0.82	562	0.66
Sciaenidae	1129	0.84	1416	0.76
Soleidae	815	0.99	64	0.93

ORGANISM RETURN DISCHARGE STATION

	(n)	DAY	(n)	NIGHT
Clupeidae	2	--	--	--
Engraulidae	1140	0.91	219	0.95
Sciaenidae	881	0.90	1014	0.71
Soleidae	817	0.99	60	0.92

CONTROL STATION

	(n)	DAY	(n)	NIGHT
Clupeidae	--	--	--	--
Engraulidae	988	0.98	398	0.99
Sciaenidae	357	0.92	555	0.89
Soleidae	--	--	4	0.80

Table 12. Comparison of percent initial and latent (48 hrs) survival of selected taxa between sampling sites at Big Bend Power Station.

Taxa	(% SURVIVAL)					
	Control		Screenwash		ORD	
	Initial	Latent	Initial	Latent	Initial	Latent
EGGS:						
Engraulidae	72	51	48	45	29	44
Sciaenidae	72	56	63	44	56	42
FISH LARVAE:						
Engraulidae	16	59	16	68	58	65
Sciaenidae	85	61	61	63	56	66
ZOEAE:						
Caridea	65	88	72	67	70	66
Xanthidae	88	85	93	80	90	71
Pinnotheridae	77	74	89	71	83	65

Table 13. Comparison of percent initial and latent (48 hrs) survival of prototype and full-scale test results.

	INITIAL % SURVIVAL			LATENT (48 HR) % SURVIVAL		
	<u>Full-Scale</u>			<u>Full-Scale</u>		
	Prototype	Screen-wash	ORD	Prototype	Screen-wash	ORD
<u>INVERTEBRATES:</u>						
Caridea	94	72	70	85	67	66
Xanthidae	99	93	90	96	80	71
Pinnotheridae	100	99	83	92	71	65
<u>FISH LARVAE:</u>						
<u>Anchoa mitchilli</u>	2	16	58	22	68	65
Sciaenidae	19	61	56	11	63	66
<u>FISH EGGS:</u>						
<u>Anchoa mitchilli</u>	43	48	29	80	82	91
Sciaenidae	75	63	56	95	84	91

DISCUSSION

Based on the survivability results of the 1980 prototype study, it was concluded that the installation of fine-mesh screening at the Big Bend Power Station Units 3 and 4 represented a viable technology which would result in lowered entrainment losses. Those results showed a high initial survival and hatchability of fish eggs; a high initial and latent survival of invertebrate taxa; and a relatively low initial and latent survivability of fish larvae, among both test and control organisms. Results of the 1985-86 full-scale study showed similar initial and latent survival and hatchability of fish eggs; a high initial and latent survival of invertebrate taxa; and a lower initial and latent survival of fish larvae (Table 13). However, the full-scale study exhibited higher initial and latent survivability of fish larvae than the prototype study. It is reasonable to assume that these differences are related to the sampling techniques used for the two studies.

It can also be seen from the results of both studies that there are natural mortalities in the organisms being studied. Survivability of organisms from the control stations showed little difference between those from the test stations. As would be expected, higher mortalities were seen at the ORD station during the full-scale study. However, these mortalities were still within expected limits which were predicted by the prototype study.

The results of the full-scale study show that the use of fine mesh screens is a viable solution to reduce entrainment. The prototype study provided the necessary information to optimize the fine mesh screen hydraulic characteristics and organism return discharge.

POWER PLANT INTAKE EFFECTS ASSESSMENT:
WHERE'S THE DRIFT?
REVIEW PAPER

J.S. Mattice
Electric Power Research Institute
Palo Alto, California

ABSTRACT

"Drift" appears to characterize the assessment of intake effects at hydroelectric and thermal electric power plants in two ways. First, intake effects stem from drift. Organisms that drift near power plant intakes are subject to impingement and the thermal, chemical, and physical stresses of entrainment. These stresses can affect individuals such that effects are translated to the population, community, and ecosystem level of organization. Although assessments can be made at any of these levels, ecosystem-level assessment is the ultimate goal. However, industry-regulatory interactions now appear most limited by abilities to conduct population level assessments, specifically by the need to define and quantify density-dependent and density-independent mechanisms that control population response. Second, the science of assessment appears to be drifting. The need for study of population control mechanisms is clear, but little concerted effort is focused on these mechanisms. Without such a focus, the drift will remain in the science of assessment, and how much of the drift of organisms into power plant intakes needs to be prevented will remain a question.

INTRODUCTION

Few subjects have received as much scrutiny over the last fifteen years as power plant intake effects. Scientists and engineers have produced a formidable number of papers on the subject. Within the constraints placed on this presentation, there were thus two possibilities: 1) select one small aspect of intake effects and deal with it exhaustively; or 2) present an overview of the whole subject with an emphasis on conclusions. This is an overview. The focus is on assessment, because every utility is facing or will face this task in the near future. This review begins with an opinion, examines the current state-of-the-art to support that opinion, and concludes with a recommendation. As such, it can be viewed as a position paper with elements of a review.

The answer to the question in the title of this paper (where is the drift?) is the basis for this presentation. The word "drift" has over 30 meanings or synonyms, according to Webster's New Twentieth Century Dictionary, Unabridged. Each of these meanings, or synonyms, implies a lack of control over directional movement. Drifting organisms are susceptible to the thermal, chemical, and/or physical effects of entrainment or to impingement on intake screens. Less recognized is the relevance of the concept of drift to the science of assessment itself. My opinion is that the major problem of power plant intake assessment has been identified. This problem is the current inability to quantify population control mechanisms, both density-independent and density-dependent (compensatory). However, because there has been no concerted effort by either the industry or regulatory communities to identify and quantify these mechanisms, we are not in control of the progress of assessment: the science is adrift. It is this drift and the research that is needed to end it that constitute the focus of this presentation.

The thesis of this presentation, then, is that the utility industry and regulatory agency communities need to focus research on quantifying population control mechanisms. I will attempt to direct attention to the need for this research by dealing with three questions: what are we trying to do; how well are we doing it; and why aren't we doing it better? The answers may seem to be belaboring the obvious.

More likely, they will be highly contestable. If so, this presentation will have successfully stimulated discussion. However, my primary intent here is to provide a convincing argument that research on population control mechanisms is a necessity --not a luxury-- if we are to improve the capability to reliably assess effects of power plant impacts. Consideration of the three questions is directed toward supporting this thesis.

WHAT ARE WE TRYING TO DO?

Most decisions about whether an impact is allowable or a mitigation is sufficient are socio-political. These decisions depend on how much society is willing to pay either in dollars for electricity (a utility industry perspective) or in maintenance of desirable environmental conditions. Three laws and their amendments direct decision making with respect to power plants: the National Environmental Policy Act; the Federal Water Pollution Control Act; and the Federal Power Act. The last is applicable to hydropower development and operation. With the recent focus on effects of hydropower generation, it is important to include turbine mortality with impingement and entrainment as intake concerns for the utility industry. Ecologists' input to decisions are most often through the impact assessments dictated by one or more of these three laws.

There is room for interpretation about how the goals of the laws are to be fulfilled. Each of the laws includes its rough equivalent to "maintenance of a balanced, indigenous population of shellfish, fish and wildlife". It seems clear that each law is aimed at balancing the benefits and costs of power generation and maintenance of a viable environment (Milburn and Ginsberg 1978; Springer and Morgan 1986; Brown 1986) with respect to intake and other effects. However, assessment and environmental protection can be viewed at a number of biological-ecological levels of organization (Figure 1). The format shown borrows elements from Coutant (1974) and Mar et al. (1985). The direction of the arrows indicates a reductionist philosophy, suggesting that impact assessment at the individual level is needed before assessment at higher levels of organization is possible. This is reasonable but not

universally accepted; others would argue that ecosystem function is the important criterion. However, if there is agreement that decisions regarding development and environmental protection are socio-political, then ecosystem function per se does not appear sufficient: it does matter to the public, for example, whether fish protein is in the form of carp or trout. The anthropocentricity of societal values may require multiple-level assessment even if the philosophy of assessment is holistic. There are few who would argue against the desirability of including total ecosystem costs in assessment, but many would argue that this is unfeasible. Thus, points of difference between utility and regulatory biologists have generally evolved into questions about the type of information that is needed to demonstrate best the balance between benefits and costs of generation.

HOW WELL ARE WE DOING?

The discussion of this question is subdivided according to the different levels at which assessments can be conducted (Figure 1). Although answers are provided for each level, they are subjective and therefore open to debate.

Exposure

Exposure does not appear to be a very useful concept with respect to impingement. This is partly due to the problem of defining impingement exposure. Are all fish found near the intake exposed to impingement? Probably, this is not the case, because some species collected in the area of intakes rarely or never appear on the screens (Loar et al. 1981; Logan and Kleinstreuer 1981). However, authors of both the cited papers found that, for certain species, distribution and abundance in the water body were important for estimating impingement. Are fish that contact the screens the exposed fraction of the population? We know that fish can contact the screens and escape at least momentarily. But does initial escape represent only a temporary reprieve from impingement? Or does the individual truly escape. Do the live fish found on the screens when they are raised represent the exposed group, or are they simply in the interim stage between impingement and the effect

of impingement (death)? The point is that there is not yet a clear definition of impingement exposure, much less a means to estimate it.

Entrainment exposures at thermal electrical power plants have all been based on the assumption that exposure is proportional to the plant flow relative to total water body flow or volume (Englert et al. 1976; Goodyear 1977; Boreman et al. 1981). In some cases, a correction factor has been added to account for distribution of entrainable organisms in the water body. Hydrodynamic models also appear to require correction factors to account for behavioral mechanisms that allow limited independence from net water body flows (Swartzman et al. 1977). The assumption that exposure to entrainment is proportional to flow and plankton distribution does not appear illogical. However, the patchiness of plankton distribution and the inability to sample continuously either near the intake and/or within the plant suggest that entrainment exposure estimates will not be verifiable.

Estimation of exposure to entrainment at hydro-electric plants appears less tractable than for thermal electric power plants. Most of the concern about exposure to turbine mortality has been for anadromous fish (e.g., salmon and American shad). Exposed fish are either fairly large, immature stages or adults. In either case, they are motile. Thus, it seems doubtful that exposure is simply proportional to the ratio of turbine water flow to total water flow past the plant for these species. In addition, problems involved in obtaining representative samples of fish in the flows through and around the turbines at most hydro-electric sites make it difficult to measure exposure or to validate predictions based on relative flows. Predictions of exposure of eggs and relatively non-motile larvae in the water column above hydroelectric sites appear more reliable, but sampling problems may be as formidable as for older life stages. Thus, exposure to entrainment probably cannot be predicted or measured very accurately at most dams.

Individual Response

Progress toward realistic prediction of impingement mortality at power plants has been slow. Impingement models have been constructed, but most are based on assumptions that are known to be unrealistic (see

Logan and Kleinstreuer 1981 and references therein). Other assessment models utilize impingement coefficients that are of questionable generality. Consensus supports fish distribution, swim speed and effects of temperature on swim speed, and overall fish fitness as parameters important for predicting impingement. Predictions using a model including these parameters (Logan and Kleinstreuer 1981), however, ranged from 20 to 548% and 35 to 143% of actual numbers impinged for threadfin and gizzard shad, respectively. The range of these differences suggests that, as yet, we have not identified all of the parameters important in determining impingement. One such parameter is differential flow across individual screens of an intake (McLean et al. 1982). Hydrodynamics in the intake areas are not simple, thus, predictions of impingement mortality must be viewed with healthy skepticism.

Substantially more progress has been made toward measuring impingement. Spatial and temporal designs have been developed to solve the problems inherent in predicting total yearly impingement from limited sampling at operating power plants (Murarka et al. 1978). If these sampling designs are followed, confidence intervals can be calculated for the impingement estimates.

Mortality of organisms (algae, zooplankton, ichthyoplankton) exposed to entrainment at thermal electric power plants is both predictable and measurable. Studies with condenser simulators (Cada et al. 1981; Jinks et al. 1978, 1981; Poje et al. 1982) and at operating power plants (Jinks et al. 1978, 1981) have shown that mortality of entrained organisms depends on the physical (thermal, mechanical) and chemical (biocides) stresses encountered. Results in the laboratory show relatively good agreement with those observed at power plants as long as levels and time courses of stress or exposure at the power plant are carefully duplicated (Jinks et al. 1978, 1981). In addition, sampling designs (Boreman and Goodyear 1981) and collection techniques (McGroddy and Wyman 1977; Jinks et al. 1981) are sufficiently advanced to provide accurate measures of entrainment mortality. Thus, entrainment mortality at thermal electric power plants can be predicted at new or operating power plants, or it can be measured once a plant is on-line.

Prediction and measurement of mortality of fish entrained at hydroelectric plants are more difficult than at thermal electric plants. Effects are thought to be mediated via pressure changes (e.g., cavitation), shear forces, and contact with conduits or turbines (Ruggles and Collins 1981). However, it has not been possible to relate types of physical injuries specifically to one of these sources. System changes also generally affect more than one of these mortality sources, so it is difficult to isolate their contributions to mortality. Add to this the fact that system size and the relative size of the system to that of the fish are important determinations of mortality, and it becomes clear that predictions must be based on studies at full-scale facilities (Turbak et al. 1981). Unfortunately, the large volumes of water passing through most hydroelectric plants make full-scale studies difficult at many sites. Full-scale studies have yielded wide ranges of mortality estimates (Ruggles and Collins 1981). In some cases the differences can be semi-quantitatively related to system design or operation, but truly quantitative conclusions are subject to valid criticism. Prediction of effects of hydroelectric plant entrainment thus seems possible only in gross terms. Even measurement is probably feasible only at a small percentage of operating plants.

Sublethal or indirect effects of entrainment at hydroelectric or thermal electric plants are thought to occur but are largely unquantified. Post-exposure observations of impingement or entrainment effects in the laboratory or at power plants have not extended past a few days, at most. This obviates observation of possible longer-term sublethal effects. Laboratory studies have shown that hungry predators selectively prey on stressed fish when stressed and unstressed prey fish are presented together (Coutant 1973). Observations that fishermen congregate near the discharges of both hydroelectric and thermal electric power plants suggest that predatory game fish are actively feeding in those areas, but whether they are feeding on dead, moribund, or stressed individuals is moot. Quantification of secondary effects such as increased predation on stressed fish may not be amenable to field investigation.

Population Response

Methods for estimating population response to intake effects can be divided into three classes depending on the level of the population for which predictions are made. Young-of-the-year (Y-O-Y) models are designed to predict the loss of the portion of population produced during a given year. Adult population models are designed to predict the effect of Y-O-Y losses on the population that would exist when the Y-O-Y class becomes vulnerable to sport or commercial fishing. Long-term models are designed to predict cumulative effects on the population of consecutive yearly losses of Y-O-Y fish at power plants.

Young-of-the-year models include those of Goodyear (1977), McFadden and Lawler (1977), Boreman et al. (1981), Lawler et al. (1981), Barnthouse et al. (1982) and others. The simplest of these are essentially "Individual Response" models (see above), because natural mortality is not included in the analysis. A second set of these models does include natural mortality rates in the Y-O-Y impact predictions, but natural mortality rates are assumed not to be affected by power plant included mortality. In other words, these models are all based on the assumption that density-dependent (compensatory) regulation is not operative on the Y-O-Y life stages. Such an assumption must be judged as conservative, surely protecting or over-protecting the year class of fish, because:

- 1) compensation is generally thought to operate during the early life stages (Science Applications, Inc. 1982);
- 2) fish populations have demonstrated the ability to maintain population stability in the face of heavy fishing mortality (McFadden 1977); and
- 3) model studies have shown that losses of Y-O-Y fish, even without compensation, are substantially less severe than equivalent losses of reproductive and fishable adults (Christensen 1985).

A third set of these models includes effects of power plant mortalities on natural mortality by including density-dependent terms in the model formulations. These models seem potentially more realistic, but both the

forms and levels of the compensatory relations have been subjects of great controversy (Christensen et al. 1981).

This controversy has raised even more questions about impact predictions based on Y-O-Y models. Such models are not intended to predict long term population responses. Lack of consideration of cumulative yearly losses of Y-O-Y fish, though, throws doubt into the cost-effectiveness of mitigation measures based on Y-O-Y models. Estimates based on these models will not necessarily be indicative of the best choices for society, because it is not clear that these Y-O-Y losses have any relationship to long term fish population changes. Nevertheless, Y-O-Y models have probably played a larger role in socio-political decisions than the other types of population models.

Adult population models include those developed by Horst (1975), Goodyear (1978), and Rago (1984). These models translate losses of Y-O-Y to adult fish by using a number of different, sometimes complex, conversion factors. Polgar et al. (1981) extended this type of projection to an "ecosystem type" analysis by conducting such extrapolations for multiple populations in a single ecosystem. Regardless of the factor(s) used for extrapolation, each of these analyses has been based on the assumption that natural mortality is unaffected by power plant induced mortality. As with the Y-O-Y models, this assumption precludes consideration of compensatory responses in the predicted effects. Cumulative yearly losses of Y-O-Y also are not included, so the actual effect may be higher or lower than predicted by these adult population models.

Long-term assessment techniques include parent-progeny (Sissenwine et al. 1974; Christensen et al. 1977; Lawler and Englert 1981; Lawler et al. 1981; Lawler 1984; Crecco 1985; and Lorda et al. 1987), stock-assessment (Jensen 1982; Jensen et al. 1982; Stanford et al. 1982; MacCall et al. 1983), and age-structure (Van Winkle et al. 1974; Christensen et al. 1975; UEC 1975; Warsh et al. 1975; Englert et al. 1976; Lorda et al. in press; Saila et al. in press) models formulated to predict multi-generation population effects. Each of these models includes one or more density-dependent functions, required to counter impact-produced extinction.

None of these models has been validated, and thus predictions based on them are subject to substantial controversy. Year-to-year variation in density-independent environmental parameters and lack of age-structured historical data obscure the relative importance and quantification of density-dependent (compensatory) relationships in the parent-progeny and stock-assessment models. Lack of agreement about quantification of specific density-dependent relationships in the more mechanistic age-structure models has made predictions based on them controversial. Thus, long-term population predictions of impact have made only a limited contribution to decisions about siting and operation of power plants.

Community Response

It has not been uncommon to deal with power plant effects at the community level. For instance, effects of entrainment on mortality of zooplankton and production of phytoplankton are more often dealt with as group responses than as responses of individual species of these groups.

Diversity indices (Lawler et al. 1981) have also been useful in some impact assessments, although under certain circumstances these appear as likely to be measures of direct individual or population responses as true community responses. Other contributions to impact assessment at the community level have been derived from cluster analysis, ordination, and niche breadth/species packing.

However, community simulation models *per se* do not appear to have been developed for assessment of intake effects. This is probably due to benefit-cost considerations and to the realities of assessment. Community analyses must deal with specific interpopulation relationships, which have proven difficult to identify and quantify. For this reason, investigators willing to make the effort to include these interactions in assessments seem to have made the additional step to ecosystem analysis. On the other hand, power plant impacts are thought to mimic abiotic effects. Strictly defined, community level analyses thus ignore some of the interactions of most interest in impact assessments. It is not surprising, then, that power plant impact models are generally available

for population or ecosystem assessments, but not for community assessments.

Ecosystem Response

Applications of ecosystem models for impact analyses which are useful for regulatory/industry purposes have been rare. The reasons are diverse but probably relate to cost, effort, and uncertainty of predictions. All of these tend to increase as level of analysis proceeds from individual to ecosystem (Figure 2). Extrapolation of effects on individuals to the ecosystem level of organization requires identification of the structure of populations and communities and quantification of their functional relations in response to both biotic and abiotic conditions. At the "simplest" level, this can be accomplished with trophic level (food web) models combined with feedback loops via decomposition and nutrient regeneration. There are problems with assessments of this type (Pikitch et al. 1978), however, especially if specific organisms or organism groups are of particular interest. Conversely, attempts to define and quantify all of the relationships in an ecosystem run up against information gaps at the population and community levels of organization. Given the complexity of ecosystem structure, such analyses become computationally unwieldy, if not impossible. Thus, ecosystem analyses have all involved some aggregation into groups. But combining leads to loss of information and reality. The ecosystem models summarized below as examples involve various levels of aggregation to facilitate analysis.

Kemp (1981) used a trophic dynamic model to compare the regional energy costs of once-through and closed-cycle cooling of a third power plant added at a site where two units already were operating. The model was a simplified simulation of biomass standing crops and transfers in a complex ecosystem. Transfers were combined into two food chains depending on the food source (plankton, detritus) of the primary consumers. The model was calibrated using data from control areas, then tested by comparing model predictions under conditions caused by the two existing plants with data collected near those power plant discharges. Agreement was "relatively close" so the model was used to predict changes

expected from addition of the third plant. Impacts were translated to photosynthetic energy equivalents using biomass transfer efficiencies between trophic levels and biomass to energy conversions. Earlier versions of this analysis were based on using total community metabolism as a proxy for energy flow (McKellar and Smith 1981). Ecosystem energy losses using a once-through cooling mode were then compared to energy and ecosystem costs of building and operating mechanical draft cooling towers to mitigate the effects of once-through cooling. The energy costs of mitigation were estimated to be about 15 to 20 times the energy losses of the ecosystem impacts. This type of analysis, however, assumes that energy is equivalent whether it is in the form of "game" or "trash" fish --an assumption that is not realistic.

Researchers at the Illinois Natural History Survey (ILHS 1979a, b, c, 1980) in cooperation with Commonwealth Edison Company staff, constructed a Cooling Lake Ecosystem Model (CLEM) as part of an effort to assess effects of operation on the Kincaid Generating Station on Lake Sangchris in Illinois. A physical model (TEMP) was used to describe water temperatures and volumes in four areas of the cooling reservoir based on meteorological and plant operational conditions. The ecosystem model (CLEM) consists of a series of submodels, which describe phosphorus, detritus, phytoplankton, periphyton, macrophytes, zooplankton, benthos, and fish dynamics. Each of the biological submodels is composed of a series of differential equations describing physiological processes. These included food intake, digestion, respiration, excretion, reproduction, growth, and mortality as functions of water temperature. Mass is conserved within and between all compartments and numbers are also conserved within the fish submodel. Transfers are generally mediated via food-consumer interactions and mortality. Sport fishing is considered explicitly. The submodels include different levels of resolution. For example, the phytoplankton submodel is divided into blue-green and non-blue-green algae modules, whereas the zooplankton submodel aggregates all (and only) herbivorous species. The fish submodel treats eight age classes each of the sportfish bluegills, largemouth bass, white bass, and channel catfish, and the forage fish, gizzard shad, separately. The model allows

evaluation of different management scenarios (e.g., plant operation and fishing regulations).

The TEMP model provided reasonable fits to temperatures measured in Lake Sangchris; the CLEM was only partially evaluated using laboratory and field data. The structure of the CLEM is general enough to be applied at other cooling lakes, but this has not been attempted.

Porcella and coworkers (Porcella et al. 1983; Grieb et al. 1983; Porcella et al. 1986) developed an Ecological Assessment Model (EAM) to predict impacts of power plants on cooling lakes. The model is a trophic dynamics model that handles biomass (dry weight) transfers and element cycling. The model can be coupled to a hydrodynamics model to predict chemical conditions at several depth strata. Trophic level analyses can include several groups of species. For example, green algae, diatoms, blue-green algae, and "others" were used to represent primary producers in the Lake Norman validation studies.

Model verification studies were conducted at Lake Norman, a cooling reservoir in North Carolina. The model was calibrated using one set of data from the reservoir, then evaluated using a separate set covering five years of measurements. Predictions of temperature, dissolved oxygen, pH, and silicate were within 10 to 20% of those measured. Nitrogen, phosphorus, phytoplankton, and zooplankton predictions were within 40 to 100% of those measured. The variations in these predictions are similar to those found for other ecosystem models and parallel the accuracy of field methods used to measure those variables. Fish biomass data were too imprecise to evaluate predicted values, but the cycles of fish biomass qualitatively matched expectations. Although further testing is required for validation, this model appears promising.

Ecosystem models inherently include some compensatory functions in their formulation. As long as these models include mass balance structures, loss of individuals within a trophic level will result in increased growth of the remaining individuals at that level because of the larger per capita availability of food. However, other compensatory relationships, including disease and parasitism, are not explicitly included in these models. These include disease and parasitism. Other

density-dependent (compensatory) trophic transfer rate changes are (or can be) included in ecosystem models. For some species, these rate changes are based on empirical data; for others the rate changes are based on generalization of data from studies on other species. The model of Porcella et al. (1986), for example includes a mix of species-specific and generic relationships to estimate effects of growth on predation. The data base for quantification of such relationships is the same as that which limits population level assessments -- definition and quantification of population interaction limits both population and ecosystem level assessments.

In other words, the ecosystem models are limited in predictive capability by problems similar to those that limit success of population models. Definition and quantification of complex relationships within and between individual species populations are necessary for accurate assessment of ecosystem impacts of power plants.

WHY AREN'T WE DOING BETTER?

There is no simple answer to this question. Development of new or improvement of existing methods of measurement and prediction at all levels of assessment from exposure to ecosystem (Figure 1) will improve power plant impact assessments. For example, improving methods of fish population sampling would improve assessments at all levels. Assessment of effects on all components of the ecosystem is the ultimate goal. However, progress toward that goal can be hastened by focusing on the level or issue that most limits the usefulness or acceptance of current assessments. For assessment of effects at the individual level, existing methods appear adequate for use at steam electric plants but need improvement to ensure reasonable accuracy at hydroelectric plants. At the other extreme, improvement of ecosystem-level assessments will at least partly involve increased consideration of community and population relationships. Thus, regardless of the initial focus, it appears that the current question of most interest is: why aren't we doing better at assessing impacts of power plants on populations?

One way to approach the answer to this more restricted question is to consider an example. The "Hudson River Controversy" (Barnthouse et

al. 1984) is an excellent example, because of the extensive documentation of the scientific analyses, their evolution, and their contribution to the ultimate socio-political decision (see Swartzman et al. 1977; Christensen et al. 1981; Barnthouse et al. 1984 and references therein).

In brief, the Hudson River utility companies agreed, amongst other things, to flow reductions and scheduled outages during certain periods of the year in return for maintaining once-through cooling. Flow reductions and outages were designed to protect sensitive life stages only during those seasons. The basis for the negotiated settlement was a series of comparisons of conditional mortalities resulting from entrainment of fish eggs, larvae and juveniles under several different cooling water flow scenarios. These Y-O-Y models (see earlier section) included natural mortality, but no density-dependent (compensatory) functions. Cumulative effects of yearly losses also cannot be considered with these models. The reason that these models were used was because the two sides could not agree on the level of density-dependent regulation in fish populations in the Hudson, or which life stages were affected by these controls. Without such agreement, long-term projections of population response (e.g., for striped bass) remained controversial. In other words, because density-dependent population regulation could not be demonstrated or quantified objectively, mitigation actions were based on assessment at the population level, but in a manner that included only annual, not cumulative, impacts.

Whether the decision was wise or foolhardy has not been determined, but the basis for that decision indicates a major deficiency in the science of impact assessment. Because density-dependent mechanisms could not be included in the impact assessment, the cumulative yearly impacts on Y-O-Y were ignored. Whether the former would ameliorate the latter is open to question. Until density-dependent mechanisms can be quantified, there can be no scientifically objective conclusion. It should be clear that, until the ability to determine the level at which a population can sustain itself in the face of anthropogenic sources of mortality is developed, predictions of the environmental impacts of power plant operation will be of questionable accuracy. We can only hope that potential errors cancel, so that society

does not incur too much in the way of unneeded economic or ecological costs.

The science of power plant impact assessment is drifting, and the problem is apparent. Identification and quantification of density-dependent (and density-independent) population controls are needed to realistically extend assessments to populations. Until a concentrated effort is directed toward quantifying population control mechanisms, science will not control the assessment process and provide information to support optimal socio-political decisions about the benefits and costs of power generation. It is time to make this effort; otherwise, like fish eggs and larvae in front of power plant intake screens, we will continue to drift on into more troublesome areas.

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IMPACT MITIGATION AND ENVIRONMENTAL ENHANCEMENT:
IT DOES WORK

J. Ross Wilcox
Environmental Affairs
Florida Power & Light Company
Juno Beach, FL 33408

ABSTRACT

Florida Power & Light Company is an active participant in impact mitigation and environmental enhancement with more than ten years of experience. The objective of this impact mitigation and environmental enhancement is to foster the compatibility of wildlife and the production of electricity. Tangible benefits have accrued to both wildlife and FPL. The primary focus of this work has been on federally designated endangered and threatened species, but the efforts have expanded to migratory species, state protected species, and habitat conservation. Examples of impact mitigation and environmental enhancement are reviewed for the West Indian manatee, three species of sea turtles, the American crocodile, the southern bald eagle, and the Barley Barber Swamp.

INTRODUCTION

Florida Power & Light Company (FPL) is the fifth largest investor owned utility in the U.S. and serves more than six million people in a service area that includes about one half the geographic area of Florida. Approximately 75% of FPL's generation is used by residential and commercial customers, in contrast to other areas of the country, where industrial customers dominate. Within our service area, which encompasses all or part of 35 counties, FPL operates 11 power plants. Power is distributed via 63,000 km of high-voltage transmission lines. Approximately one-third of the generating capacity of the system is provided by nuclear energy, one-half by oil and natural gas, and the balance by interchange and purchase power.

The Company owns and manages approximately 325 km² of land in Florida. This land is located in urban and rural areas and comprises wetland and upland areas. Much of it is undeveloped and is used as a buffer zone to isolate power plants, substations, and transmission lines from agricultural and residential interests. These undeveloped buffer zones serve as wildlife habitat for many species, both common and protected.

Florida's unique environment of habitat and climate presents FPL with some interesting challenges. For instance, 40% of our service territory includes wetlands, swamps and lakes; about 40% is forested, and about 20% is cropland or urbanized. The Company must build or maintain power plants, transmission lines, and substations in all these environments.

Climates in Florida show great variation also and create habitats for a diverse array of plants and animals not seen elsewhere in the continental United States. For instance, freezing to temperate conditions are common in northern Florida, such as Gainesville, and permit only certain types of plants and animals to exist. Conversely, frost rarely occurs in South Florida, such as Miami, and allows a subtropical group of plants and animals to flourish.

Many plants and animals are common and frequently seen throughout Florida. On the other hand, because of the sensitivity of Florida's habitats to water cycles and the conversion of wetland and forested habitats to serve the agricultural and population needs of Florida, a

large number of plants and animals are near extinction. On the federal level, more than 60 plants and animals of Florida has a more comprehensive list that includes 541 species of concern (Wood, 1986). During the production and transmission of electricity, FPL interacts with both common and endangered species.

Florida Power & Light Co. is an active participant in impact mitigation and environmental enhancement with more than ten years of experience. The objective of the efforts is to foster the compatibility of wildlife and the production of electricity (Wilcox, 1979, 1980; Morgenthaler, 1987). The primary focus of this work has been on federally designated endangered and threatened species, but the efforts have expanded to migratory species, state protected species, and habitat conservation. Tangible benefits have accrued to both wildlife and FPL and will be reviewed in this paper.

DISCUSSION

West Indian Manatee

The West Indian manatee is an endangered marine mammal that FPL has interacted with extensively over the last 10 years. Because of the intolerance of the species to water temperatures below 19°C, the animal has sought the once-through thermal discharges of five FPL plants along the east and west coasts of Florida. In regulatory proceedings before the Environmental Protection Agency (EPA), this species constituted a key factor in the permitting review. Because of the Endangered Species Act, EPA was required to consult with the U.S. Fish and Wildlife Service (USFWS) to insure that the "permitted activities would not likely jeopardize the continued existence of an endangered species".

Utilization of the thermal discharge by manatees was one of the factors that influenced EPA to let FPL continue operating in the once-through cooling mode, instead of off stream by adding cooling towers. Taken to worst case extremes, this decision saved the Company and its rate payers approximately \$300 million at two power plants. As part of the consultation in this regulatory process, FPL and USFWS decided that not enough was known about the interaction of manatees and power plants. To mitigate any impact on the manatee from this regulatory decision, FPL

agreed to conduct two manatee programs: an aerial census program and a public awareness program. The aerial program has continued on a yearly basis since 1977 (Reynolds and Wilcox, 1986).

Benefits have accrued to the resource and the company from these aerial surveys. The primary benefit for the manatee is scientific data on which to base conservation and management efforts. These counts were, in part, the basis for raising the official estimate of animals in the population and identifying specific areas for boat speed limits. Tangible and intangible benefits to the company include: national and state recognition for FPL's manatee conservation efforts, information to help schedule maintenance outages of units without impact to manatees, and scientific data to support permit renewal or issuance.

As another facet of mitigation, the Company agreed to initiate a public awareness program for manatees. FPL has sponsored or conducted these public awareness programs yearly since 1977 directly reaching over 10,000 people. As an offshoot of this public awareness effort, FPL has distributed over 530,000 pieces of literature on manatee conservation to the public.

Tangible and intangible benefits have also accrued to the resource and the Company from the literature and the public awareness programs. For instance, the public is more aware of the plight of the manatee, and many people will go out of their way to help an injured manatee or will respect boat speed limits in manatee sanctuaries. Each piece of literature distributed has FPL's name on it, even though it is unobtrusive. However, it is still a tangible expression to the public and FPL's rate payer about FPL's commitment to the environment.

Personnel of the Riviera plant located near West Palm Beach have developed an innovative method to provide warm-water effluent to manatees. Because the two units that have traditionally attracted the animal are no longer operational, plant personnel devised a method to siphon heated effluent from two other operational units into a basin that the manatee favors. Based on the successful demonstration of a temporary and small siphon in 1984, personnel installed two 30 cm diameter siphons, one for each operational unit at the plant. The siphons intercept a small portion (1%) of the heated effluent from the units before it is

discharged into Lake Worth. Thus, a small but sufficient amount of heated effluent is diverted from an area not favored by the manatees to a basin adjacent to the plant that the animals have traditionally favored.

This mitigation scheme has been successful because subsequent aerial surveys documented heavy use of this basin with more than 200 manatees being observed during several cold fronts. Since each unit has a siphon and one or both units are operating almost continuously and always during cold weather, manatees will have a continuing source of warm water to assist their survival.

Another example of impact mitigation for manatees occurred at FPL's Ft. Myers plant in January 1985. Because the units were not operating for various reasons, the Company was faced with a situation where no warm effluent was available to aggregating manatees that had traditionally been conditioned to this discharge (Packard et al., 1985). A paradox arose: does FPL operate a generating unit just to keep manatees warm?

As ambient water temperature cooled during this January and approached levels that would stress manatees, senior management, state and federal officials, field biologists, and meteorologists were consulted. Due to a severe cold front, Unit 1 was dispatched to meet record electrical demand for three days. Over 100 manatees that had aggregated in a deep but warmer pocket of water nearby immediately found the warm-water discharge and stayed in the discharge canal. After the dissipation of the cold front and as electrical demand abated, the Company faced another paradox: does the unit shut down for economic reasons and subject more than 100 manatees to stressful if not harmful water temperatures?

After much discussion with senior management and with their agreement, the unit was operated for 11 days out of economics (e.g., FPL could have produced electricity cheaper at another plant) during the months of January and February. The sole reason to operate the plant was to enhance the survival of manatees.

Senior management consented to operate the plant during this critical period only on the condition that they would not be asked to do

it again. Therefore, an alternative method to provide a thermal discharge to manatees was needed.

After much discussion and evaluation, FPL decided to tap ground water under artesian pressure. A series of three 25 cm diameter wells were drilled up to 260 m depth. The ground water temperature was 23°C, and with a pumped flow of 3800 l/min, an area of approximately 2 hectares would be provided as a warm-water refuge for the manatees.

Even though the cost of the manatee protection wells was nearly \$500,000, there was a net benefit to the Company, because the power plants could now operate on economic dispatch instead of "manatee dispatch". The benefit to the manatees is the maintenance of a traditional refuge that more than 300 animals seek.

Sea Turtles

Florida Power & Light Company has interacted with endangered or threatened sea turtles over the last ten years, primarily at the St. Lucie plant on the eastern coast of central Florida. The species include: green (endangered in Florida); leatherback (endangered); Kemp's ridley (endangered); Hawksbill (endangered); and the loggerhead (threatened). The green, leatherback, and loggerhead nest on FPL beaches, while the Hawksbill and Kemp's ridley only occur in the coastal waters of the area.

The Company has actively participated in mitigation and environmental enhancement for sea turtles with a variety of programs. Some programs have been driven by the regulatory process, and some have been voluntary. Because of dual regulatory jurisdiction for sea turtles, the National Marine Fisheries Service (NMFS), which has jurisdiction over sea turtles in the water, and the USFWS, which has jurisdiction over sea turtles while on beaches, extensive coordination was required. Additionally, the Nuclear Regulatory Commission and the EPA were also involved in sea turtle concerns at the St. Lucie Plant.

When the St. Lucie Unit 1 became operational in 1976, several loggerhead turtles were unexpectedly taken from the intake canal of the plant. The turtles had entered the ocean intake structures and traveled through a series of large underground culverts to an open canal, where

they became entrapped. The details of this entrapment are reviewed in Ernest et al. (1987).

For all practical purposes, the St. Lucie units operate as a turtle trap. During this process, some animals are injured or killed. As mitigation for this "take" of a protected species, FPL tags the turtles and obtains a variety of biological information, which includes: blood samples, morphometric data, past injuries, present health, and recapture data. This data base is available to all that request it and has been used extensively for federal, state, and university research programs. All this has benefited the conservation and scientific understanding of the resource.

The benefit to FPL is the ability to operate a key facility within the limits of the law and with the concurrence and understanding of federal and state agencies. The cost to physically exclude turtles from the St. Lucie plant was estimated at \$200 million for a one-time capital cost and a yearly maintenance cost of \$1 million, while the cost to remove entrapped turtles is \$500,000/year. Other forms of exclusion, including sound and visual stimuli, were investigated in a research program and were found to be impractical (O'Hara and Wilcox, 1984).

Another aspect of mitigation and environmental enhancement deals with FPL's documentation of sea turtle nesting on the beaches of the St. Lucie Plant. This program has been conducted since 1971 and is ongoing. During the turtle nesting season of May through September, FPL conducts daily nesting surveys on 36 km of beaches to document nest numbers and distribution. Details and results of this program are documented in Gallagher et al. (1972), Worth and Smith (1976), Williams-Walls et al. (1983), and Proffitt et al. (1986).

This program has generated one of the most comprehensive and extensive data bases on nesting sea turtles in Florida. Benefits to the resource include knowledge for each species of sea turtle concerning intra-seasonal nesting intervals; inter-seasonal nesting cycles; nest and hatchling success; predation; and natural and human impacts on nesting. Benefits to the Company include demonstration that past construction impacts on the beach have been temporary and not permanent and that the thermal discharges from the plants have no impact on the nesting of sea

turtles or the survival of their hatchlings (O'Hara, 1980). This knowledge has expedited the renewal or issuance of federal and state permits.

One impact on sea turtle behavior identified in the permit review process was the effect of plant security lighting had on hatchling orientation and their return to the water. To mitigate this impact, FPL has minimized beach lighting and, where it could not be reduced, has established a light screen with vegetation to minimize this disorientation problem.

As a continuing effort to demonstrate the Company's commitment to the environment and to voluntarily mitigate any impact from beach lighting in general, FPL has produced a bumper sticker and poster to promote awareness of turtle hatchling disorientation and what the general public can do about this problem. Additionally, the company has produced and distributed a bill insert promoting this awareness of hatchling disorientation to FPL customers on the east coast of Florida. By promoting this awareness, the resource is benefited and FPL is established as having an environmental conscience.

Another form of voluntary mitigation involves the turtle walks that the Company provides as a public service. Organized and guided tours to view a female sea turtle crawl up on the beach to lay her eggs are very popular nature experiences during June and July; FPL has conducted these walks since 1983. Reservations, which are required, can be made through FPL, and the walks are conducted on FPL property in the shadow of a nuclear power plant. FPL staff conduct an orientation program about sea turtle biology and conservation prior to the actual walk. FPL produced public awareness materials on sea turtles; these materials are distributed at the walks and to the general public. Approximately 80,000 sea turtle booklets have been distributed through these and other public awareness programs.

Again, the resource is benefited because the insatiable appetite of the public to view sea turtles nesting is being channelized in a constructive and beneficial program, while FPL is demonstrating the compatibility of our facilities with sea turtles and the Company's environmental commitment.

American Crocodile

Florida Power & Light Company interaction with the endangered American crocodile dates to 1976 when hatchlings were first discovered in the cooling canal system of the Turkey Point Plant. The plant is located 60 km south of Miami and produces electricity with fossil and nuclear units. Subsequent monitoring (i.e., a form of voluntary mitigation) outside the regulatory or permitting arena established that a small but viable crocodile population inhabits the site, all of which has been designated by the USFWS as critical habitat. Details and results of this monitoring program are given in Gaby et al. (1985) and Masotti et al. (1986).

This monitoring program benefited the resource because it established the dynamics of the population, recruitment of animals into the next size class, the behavior of animals, and how they partitioned the environment. All this is useful data. These studies also benefited the Company during federal licensing activities of the nuclear units. As with many nuclear issues, intervenors use all types of strategies to prevent or delay this licensing and crocodiles became an intervention issue. However, because of the extensive data base on crocodiles developed during the monitoring program, the Company was able to easily answer any interrogatories and a \$200 million project was not blocked or delayed.

In 1984, the Company initiated a maintenance program to remove exotic trees and to clean cooling canals of accumulating spoil. To mitigate any impacts of these activities, a crocodile management plan was drafted and reviewed by all affected FPL Departments. By knowing the behavior and life requirements of the animals, a sensible maintenance schedule was established that allows the animals to nest in certain areas while allowing maintenance dredging in areas not inhabited or utilized by the animals. Thus, both the resources and the Company benefited.

As a voluntary mitigative effort, the Company developed a public awareness booklet on Florida's alligators and crocodiles. This booklet promotes a better understanding of the animals and is another example of FPL's commitment to the environment.

Southern Bald Eagle

Florida Power & Light Company has monitored nesting of the southern bald eagle around company facilities over a four-year period. The purpose of this monitoring program was to insure the compatibility of company operations or construction and the well-being of an endangered species. Details and results of the program are given in Williams-Walls (1986).

This monitoring program verified the mitigation policies established by the U.S. Fish & Wildlife Service for eagle nesting (USFWS, 1987). The service recommended two buffer zones around an active eagle's nest and the level of human activities that would not disturb the pair of eagles.

With prior knowledge of this mitigation policy, the company was able to construct a 500 kV switchyard and a tower structure for another 500 kV line that was within the outer buffer zone established by the service. Land clearing and tower erection requirements were established and incorporated in the planning and subsequent construction schedule. Activities were timed so that the eagles and their nesting activities were not impacted during the construction phase. Subsequent monitoring of the operating lines demonstrated no impact. Similarly, no impacts were observed on eagle nesting from the operation of nearby power plants. Proper timing of activities benefited the resource, and the Company was able to obtain federal and state permits to construct a switchyard and transmission facility without delay.

Barley Barber Swamp

In 1972, FPL purchased approximately 3,650 hectares of land for the Martin Power Plant and a cooling reservoir near Lake Okeechobee. During planning stages for the cooling reservoir, a unique cypress strand known as the Barley Barber Swamp was singled out as worthy of preservation. In consultation with several state agencies, approximately 160 hectares of the swamp were preserved by building the levy of the cooling reservoir around the area.

Through environmental monitoring, the uniqueness of the swamp became more and more apparent. A group of cypress trees over 400 years old were identified as well as a rookery area of the endangered woodstork. Other important species observed include southern bald eagles (endangered) and indigo snakes (threatened).

To demonstrate the compatibility of the swamp with the Martin Plant, the Company constructed a 2.2 km long boardwalk through the heart of the swamp and opened it to visitors by appointment. The boardwalk, completed in 1980, offers visitors a unique opportunity to view a pristine wetland and upland community as it may have existed before the vast land alterations and water control efforts of the last 100 years. To further enhance the functioning of this unique area, a series of management and enhancement plans were drawn up and implemented with expert help. These plans called for channelizing water into the swamp from reservoir seepage sumps to simulate sheet flow; manipulation of water levels to simulate hydroperiods for the wet and dry season; controlled burns to maintain certain areas in specific stages of ecological succession; and efforts to control exotic vegetation. The management plan recommended that several 300-500 year old Indian mounds adjacent to the swamp be preserved and checked for archeological significance.

The benefits to the resource are overwhelming. Not only have eagles and woodstorks nested intermittently in the swamp, but over 150 species of birds have been identified in or around the swamp. Species lists for mammals and reptiles are also extensive. Twenty species of ferns and six species of orchids have been identified, as well as over 200 species of wildflowers, plants and trees.

The benefits to the Company are also overwhelming. Since the opening of the boardwalk, over 12,000 visitors from 30 states and eight foreign countries have toured the swamp by appointment through FPL. These visitors have been for themselves how a power plant can operate 3 km away and yet be compatible with the resource. Many of the visitors touring the swamp are from conservation organizations, such as the Audubon Society, Sierra Club, and Native Plant Society. The swamp has proven to be so popular with these organizations that they plan seasonal

tours to observe changes in the flora and fauna. The significance of all these contacts is that our customers and the public can observe FPL's tangible concerns for the environment.

CONCLUSIONS

As the examples demonstrate, mitigation and environmental enhancement do work and benefit both the resource and private industry. Utility management may have difficulty justifying and accepting the costs for the program cited, but it is up to the utility biologist or a similar counterpart to point out the tangible and intangible benefits to the Company.

As part of FPL's Quality Improvement Program, senior management have adopted the following Corporate Vision: "During the next decade, we want to become the best managed electric utility in the United States and an excellent company overall and be recognized as such."

In 1984, the Company received the Conservation Service Award, the highest award given by the Department of Interior to a private organization, for the Company's work on endangered species. Likewise, in 1986 the Florida Audubon Society recognized the Company by presenting the Corporate Conservation Award for promoting public awareness and protection of Florida's endangered species.

This state and national recognition is evidence that Florida Power & Light Company programs are innovative, forward looking, and an accepted means to promote mitigation and environmental enhancement. Instead of fighting the system, why not use your professional skills as biologists, engineers, and managers and work with the system in order to benefit both the resource and the business community alike?

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WHEN ARE TEMPERING PUMPS AN EFFECTIVE BAT
FOR MITIGATING THERMAL IMPACTS?

Linda R. Cadman
A.F. Holland
Stephen B. Weisberg
Versar, Inc.
ESM Operations
9200 Rumsey Road
Columbia, Maryland 21045

ABSTRACT

Thermal impacts of once-through cooling are sometimes mitigated by diluting thermal discharges with ambient water. However, decreased thermal effects are gained at the expense of increased entrainment through tempering pumps. A model to quantify this trade-off was applied to empirical data collected for the Chalk Point Steam Electric Station in Maryland and tempering pumps were found to increase total mortality for representative species of fish and crabs. Sensitivity analysis of the model's assumptions and input parameter estimates substantiated the finding that tempering pumps were detrimental at Chalk Point. As a result of these analyses, the decision was made to discontinue their use. Further examination of model inputs suggest that tempering pumps would only be effective at Chalk Point under a limited set of circumstances. These analyses demonstrate the need for evaluating the ecological costs in relation to anticipated benefits for any mitigative measure.

INTRODUCTION

Section 316(a) of the Clean Water Act requires utilities to comply with state and federal thermal effluent limitations or demonstrate that discharges do not adversely affect balanced indigenous populations. One means used to meet thermal effluent limitations and reduce adverse thermal impacts is to lower the temperature of the discharge water by diluting it with unheated water. Dilution occurs before the discharge is returned to the receiving water body and is accomplished by large pumps (referred to as tempering or dilution pumps) that transport water from the intake area to the discharge area.

Tempering pumps, however, increase a facility's total water withdrawal rate. The increase in water withdrawal rate produces an increase in the number of organisms susceptible to impingement and entrainment. Therefore, decreased thermal impacts are gained at the expense of higher impingement and entrainment must be evaluated before the effectiveness of tempering pumps as a mitigative technology can be assessed.

In 1967, tempering pump operation was initiated at the Chalk Point Steam Electric Station (SES) to mitigate thermal impacts. At the time, no analysis of the adverse impacts of increased entrainment was conducted. In 1985, we developed a simple model for evaluating the trade-offs between reduced discharge temperature and increased entrainment losses associated with tempering pumps. This model was used to estimate relative increases or decreases in total plant-induced mortality due to tempering pump operation at Chalk Point.

METHODS

Study Site

The Chalk Point SES, located in the estuarine portion of the Patuxent River, Maryland, uses 30 m³ of water per second for once-through cooling. Cooling water is withdrawn from a 137-m-long intake canal and returned to the Patuxent River 2 km upstream from the plant via an 18-m-wide discharge canal (Figure 1). Intake structures are protected by a barrier net, a trash rack, and traveling screens. Organisms entrained or impinged at the plant are washed into the discharge canal. Transit time for a passive particle through the discharge canal is 2-4 hours (Academy

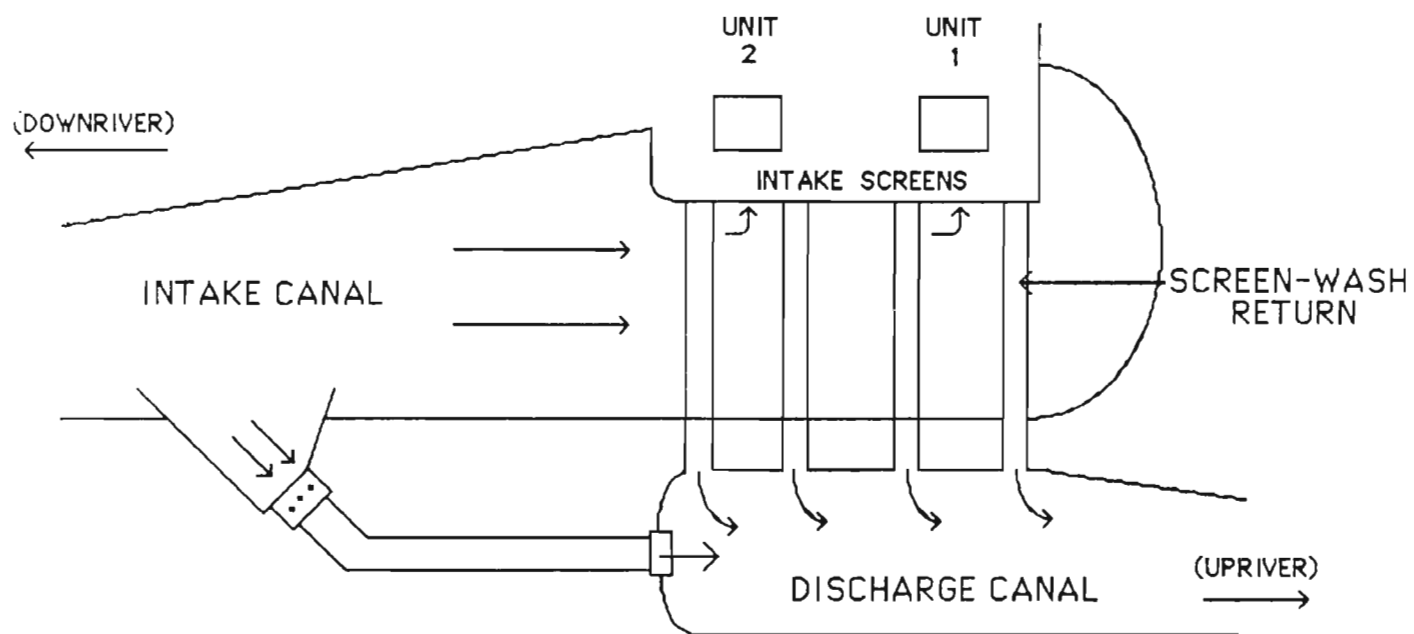


Figure 1. Top view schematic of intake and discharge systems at Chalk Point Steam Electric Station, Patuxent River, Maryland (from Hirshfield et al, 1982).

of Natural Sciences of Philadelphia, 1983). Temperature at the head of the discharge is as much as 20° C higher than the intake temperature (Martin Marietta Environmental Systems, 1985). The average ΔT during the summer is 7.4°C.

From 1969 to 1985, three tempering pumps, each with a pumping capacity of about 5 m³ sec⁻¹, were used to pump water directly from the receiving water body to the discharge canal from June to September. Operation of all three tempering pumps reduced the temperature at the head of the discharge canal by about 2.7°C and by less than 0.5°C in the nearfield (Academy of Natural Sciences of Philadelphia, 1983).

Analytical Approach

The general approach used to evaluate the effectiveness of tempering pumps involved identification of all sources of power plant-induced mortality and comparison of total mortality with and without tempering pump operation. The generalized model was of the form:

$$\begin{array}{rclcl}
 \text{Total Plant-} & & & & \text{Circulating} & & \text{Tempering} \\
 \text{Induced} & & \text{Intake Screen} & & \text{System} & & \text{Pump} \\
 \text{Mortality} & = & \text{Impingement} & + & \text{Entrainment} & + & \text{Impingement} \\
 & & \text{Mortality} & & \text{Mortality} & & \text{Mortality} \\
 & & & & & & \\
 & & \text{Tempering} & & & & \\
 & & \text{Pump} & & & & \\
 & & \text{Entrainment} & + & \text{Nearfield} & & \\
 & + & \text{Mortality} & & \text{Mortality} & &
 \end{array}$$

The effectiveness of tempering pumps at Chalk Point SES was evaluated by comparing estimates of mortality for four important fish species and blue crabs with and without tempering pump operation.

Each impingement and entrainment term includes mortality directly due to the process of impingement or entrainment plus the mortality of surviving organisms caused by thermal or chemical stress in the discharge. Nearfield mortality is defined as mortality of organisms residing in the nearfield area that is caused by thermal or chemical stress induced by the discharge effluent.

When the model was applied to Chalk Point, several terms in the model were not included because they were not altered substantially by tempering pump operation; they were, in effect, constants. These sources of mortality were unnecessary for the analysis since the objective was not to estimate total mortality but to determine the difference in

mortality between the two operating modes. The components not included in the analysis for Chalk Point were circulating system entrainment, tempering pump impingement, and discharge effects on organisms residing in the nearfield.

The term for circulating system entrainment mortality was not included because it approaches 100% due to continuous chlorination during the summer when tempering pumps are used (Martin Marietta Environmental Systems, 1985). This mortality rate is unaltered by operation of tempering pumps since entrained organisms are dead before they reach the discharge canal where the temperature reduction due to dilution occurs (Martin Marietta Environmental Systems, 1985).

Tempering pumps at Chalk Point are unscreened. Thus, tempering pump impingement mortality is zero and this term was not included in the analysis for Chalk Point. However, larger organisms that would normally be impinged were entrained through the tempering pumps.

Discharge effects in the receiving water body were not included in application of the model to the Chalk Point SES because neither chlorine nor thermal conditions in the nearfield are altered greatly by operation of the tempering pumps. Total residual chlorine (TRC) concentrations at the head of the discharge canal have historically been less than 0.20 mg/l and essentially no TRC is discharged to the receiving water body (Academy of Natural Sciences of Philadelphia, 1983). Further, hydrographic model simulations predicted that the ΔT in the nearfield is not substantially altered by tempering pump operation. The largest reduction in ΔT anywhere in the receiving water body due to tempering pumps was less than 0.5°C (Academy of Natural Sciences of Philadelphia, 1983).

After deleting the sources of mortality that are unaffected by tempering pump operation, the formula applied at Chalk Point SES was reduced to two sources:

$$\begin{array}{ccccc} \text{Plant-} & & \text{Intake Screen} & & \text{Tempering Pump} \\ \text{Induced} & & \text{Impingement} & & \text{Entrainment} \\ \text{Mortality} & = & \text{Mortality} & + & \text{Mortality} \end{array}$$

Impingement losses were calculated as the sum of immediate mortality associated with the mechanical stresses of impingement plus the additional mortality due to thermal stress when the organisms are washed into the discharge canal (Table 1). Immediate impingement losses were

Table 1. Mortality components used in the analysis of tempering pump effectiveness at Chalk Point SES

	Intake Screen Impingement Mortality		Tempering Pump Entrainment Mortality	
	Immediate	Thermal Component	Immediate	Thermal Component
Tempering Pumps On	$A \cdot B$	$A(1-B) \sum_{t=1}^5 E_t F_t$	$C \cdot D$	$C(1-D) \sum_{t=1}^5 E_t F_t$
Tempering Pumps Off	$A \cdot B$	$A(1-B) \sum_{t=1}^5 E_t G_t$	-	-

where

A = number impinged on intake screens

B = immediate impingement mortality rate

C = number entrained through the tempering pumps

D = immediate entrainment mortality rate

E_t = thermal mortality rate in temperature interval t

F_t = proportion of time the discharge temperature was in temperature interval t based on diluted discharge temperatures

G_t = proportion of time the discharge temperature was in temperature interval t based on undiluted discharge temperatures

calculated by multiplying the total number of a species impinged during June through September by the immediate impingement mortality rate for each species. Similarly, tempering pump entrainment losses were calculated as the sum of immediate mortality due to mechanical stresses associated with entrainment through tempering pumps plus the additional mortality due to thermal stress when surviving organisms are washed into the discharge canal. Tempering pump entrainment mortality was calculated as the product of the total number entrained during June through September and the immediate entrainment mortality rate. Estimates of these parameters for Chalk Point (Table 2) were taken from Hirshfield et al (1982).

The thermal mortality component of both impingement and tempering pump entrainment was computed by multiplying the number of organisms that survived entrainment and impingement by the expected mortality rate due to thermal stresses in the discharge canal. The expected thermal mortality rate was calculated using thermal tolerance data obtained from the literature. Lethal temperatures (LT) were obtained for a range of acclimation temperatures for each species evaluated. Temperatures for 0, 50, and 100 percent thermal mortality (i.e., LT₀, LT₅₀, and LT₁₀₀) were determined based on available literature estimates and values for LT₂₅ and LT₇₅ were estimated by linear interpolation. To estimate the average thermal mortality rate for the summer months, the expected percent mortality for each temperature interval (e.g., between the LT₂₅ and the LT₅₀) was weighted by the of time the discharge temperature was in that temperature interval. The frequency distribution of discharge temperature over time was determined from monitoring studies conducted in the discharge.

An organism's temperature tolerance depends on its thermal history, and therefore our model was run twice assuming different acclimation temperatures: 17°C and 32°C. Intake temperature at Chalk Point during the study period ranged from 17° to 32°C. Thermal mortality was minimized by assuming that the organisms impinged or entrained at the plant were acclimated to 32°C for the entire study period. Similarly, thermal mortality was maximized by assuming that the organisms were acclimated to 17°C. Parameter values used for estimating thermal mortality for each species and acclimation are given in Tables 3 through 6.

Table 2. Impingement and entrainment parameter values used to estimate mortality at Chalk Point SES with and without tempering pumps operating

Parameter	Blue Crab	White Perch	Striped Bass	Spot
Number impinged	180,224	11,722	4,762	25,744
Impingement mortality rate	10.0%	19.0%	33.0%	36.0%
Number entrained	80,307	76,773	3,348	37,286
Entrainment mortality rate	41.8%	69.7%	69.7%	91.0%

Table 3. Parameter values used to estimate blue crab thermal mortality at Chalk Point SES with and without tempering pumps operating

Acclimation	Temperature Interval t (°C)	Mortality Rate (E _t)	Proportion of Time Spent in Interval t	
			Pumps On (F _t)	Pumps Off (G _t)
LOW	<34.0	0.00	0.622	0.454
	34.0-35.0	0.25	0.032	0.060
	35.0-36.0	0.50	0.067	0.048
	36.0-36.5	0.75	0.047	0.035
	>36.5	1.00	0.232	0.403
HIGH	<37.0	0.00	0.819	0.632
	37.0-38.0	0.25	0.044	0.035
	38.0-39.0	0.50	0.041	0.092
	39.0-39.5	0.75	0.003	0.060
	>39.5	1.00	0.092	0.131

Table 4. Parameter values used to estimate white perch thermal mortality at Chalk Point SES with and without tempering pumps operating

Acclimation	Temperature Interval t ($^{\circ}\text{C}$)	Mortality Rate (E_t)	Proportion of Time Spent in Interval t	
			Pumps On (F_t)	Pumps Off (G_t)
LOW	<25.0	0.00	0.054	0.010
	25.0-26.5	0.25	0.079	0.016
	26.5-28.0	0.50	0.130	0.044
	28.0-29.0	0.75	0.073	0.054
	>29.0	1.00	0.664	0.876
HIGH	<34.0	0.00	0.622	0.454
	34.0-35.5	0.25	0.057	0.070
	35.5-37.0	0.50	0.140	0.108
	37.0-37.5	0.75	0.016	0.022
	>37.5	1.00	0.165	0.346

Table 5. Parameter values used to estimate striped bass thermal mortality at Chalk Point SES with and without tempering pumps operating

Acclimation	Temperature Interval t (°C)	Mortality Rate (E_t)	Proportion of Time Spent in Interval t	
			Pumps On (F_t)	Pumps Off (G_t)
LOW	<27.0	0.00	0.181	0.038
	27.0-28.0	0.25	0.083	0.032
	28.0-29.0	0.50	0.073	0.054
	29.0-30.0	0.75	0.038	0.092
	>30.0	1.00	0.625	0.784
HIGH	<33.0	0.00	0.549	0.406
	33.0-34.5	0.25	0.083	0.083
	34.5-36.0	0.50	0.089	0.073
	36.0-37.5	0.75	0.114	0.092
	>37.5	1.00	0.165	0.346

Table 6. Parameter values used to estimate spot thermal mortality at Chalk Point SES with and without tempering pumps operating

Acclimation	Temperature Interval t (°C)	Mortality Rate (E _t)	Proportion of Time Spent in Interval t	
			Pumps On (F _t)	Pumps Off (G _t)
LOW	<27.0	0.00	0.181	0.038
	27.0-29.0	0.25	0.155	0.086
	29.0-31.0	0.50	0.102	0.143
	31.0-32.0	0.75	0.051	0.070
	>32.0	1.00	0.511	0.663
HIGH	<33.0	0.00	0.549	0.406
	33.0-34.5	0.25	0.083	0.083
	34.5-36.0	0.50	0.089	0.073
	36.0-37.0	0.75	0.098	0.070
	>37.0	1.00	0.181	0.368

RESULTS AND DISCUSSION

Tempering pumps at Chalk Point SES were not only found to be ineffective, but they actually increased total mortality of fish and crabs. The estimated mortality for the four representative species was 18 to 90% higher with tempering pumps operating than without the pumps operating (Table 7).

The conclusion that tempering pumps are detrimental rather than beneficial to the Patuxent River ecosystem was substantiated by sensitivity analysis of the model to input parameters. For example, immediate impingement mortality rate can range from 0 to 100%. Analyses indicated that altering impingement mortality rate alone did not change the outcome of the model. That is, even if all organisms survived the mechanical stress of impingement, tempering pumps at Chalk Point would still increase total mortality of the representative species. Similar results were found when immediate entrainment mortality was varied between 0 and 100%, with one exception. If the blue crab mortality due directly to being entrained through the tempering pumps is very low (less than 22% for the high acclimation scenario and less than 3% for the low acclimation scenario), then tempering pumps would reduce mortality of blue crabs. However, at Chalk Point, it is unlikely that pump entrainment mortality would even be this low. The on-site estimate of immediate mortality for entrained blue crabs was 41.8% (Potomac Electric Power Company, 1984). If delayed mortality had been included in this estimate, the mortality estimate would have been even higher.

The effect of varying the ΔT reduction caused by the tempering pumps was also examined. For the representative fish, altering the ΔT up to 10.0°C did not change the outcome of the analyses (assuming that nearfield effects remain relatively unchanged). Even a 10.0°C decline in discharge temperature would not decrease thermal mortality enough to offset the increased mortality due to tempering pump entrainment. For blue crabs, however, losses would be lower if the reduction in discharge temperature resulting from pump operation was greater than 4.5°C. To reduce the discharge temperature at Chalk Point by 4.5°C would require the pumping of about 46% more dilution water than is currently pumped. This would greatly increase the number of organisms entrained through tempering pumps. Therefore, it is unlikely that any errors in the

Table 7. Estimates of mortality at Chalk Point SES with and without tempering pumps operating

Species	Acclimation	Mortality Estimate		Percent Difference
		Pumps On	Pumps Off	
Blue Crabs	Low	116,097	93,974	19.1
	High	77,957	63,593	18.4
White Perch	Low	82,048	11,179	86.4
	High	64,291	6,349	90.1
Striped Bass	Low	6,895	4,405	36.1
	High	5,303	3,124	41.1
Spot	Low	55,868	22,593	59.6
	High	49,541	17,140	65.4

parameter estimates used would alter the conclusion that tempering pumps at Chalk Point are detrimental.

At Chalk Point, the general model for estimating changes in total mortality was reduced to two components -- tempering pump entrainment and intake screen impingement. The trade-off to be considered was simply reduction in thermal mortality of impinged organisms in the discharge canal versus the additional mortality due to tempering pump entrainment. The analysis approach facilitated an evaluation of the trade-offs and the conclusion from the Chalk Point SES seems clear. Tempering pumps are not an acceptable technology for mitigating thermal effects because they actually increase the number of organisms killed by the plant.

Although tempering pumps were ineffective at Chalk Point and a similar finding was made at the Big Bend facility in Florida where operation of dilution pumps was discontinued (U.S. Environmental Protection Agency, 1981), there are situations where tempering pumps could be potentially useful for achieving compliance with thermal statutes or for mitigation of thermal impacts. For instance, tempering pumps at Chalk Point do not reduce nearfield temperatures significantly; at facilities where the ΔT in the nearfield is effectively reduced, tempering pumps are more likely to reduce total mortality. Similarly, if migration routes of anadromous fish are blocked by a thermal plume, dilution pumping may reduce the size of the plume sufficiently to allow migration. In such a circumstance, it is possible that tempering pumps might still increase plant-induced mortality, but the benefit of open migration routes may be "worth" the cost of additional mortality to another life stage or species.

Tempering pumps may also be useful for reducing cold shock associated with plant outages. When organisms are attracted to elevated temperatures in a discharge, they are subject to cold shock mortality if the plant shuts down. Dilution pumping may help reduce attraction to the discharge by reducing ΔT s in the nearfield and thus the potential for cold shock mortality.

Any proposed mitigative measure, whether it directly mitigates an impact or indirectly replaces a loss, has associated costs and benefits, and obviously both need to be considered in the recommendation and decision-making process. Historically, the approach to evaluating the

costs and benefits of a technology to be applied for environmental reasons has been to consider only the economic costs (i.e., capital, operating, and maintenance costs) and the ecological benefits (e.g., reduction in a specific impact to be mitigated). The Chalk Point analysis has shown that it is of great importance that all ecological costs also be considered. In addition, it may also be necessary to evaluate the relative value of the ecological costs and benefits to facilitate evaluation of trade-offs. Finally, it is imperative that monitoring be conducted after a technology has been applied to confirm the predicted costs and benefits.

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EVALUATION OF FISH DIVERSION EFFICIENCIES AND SURVIVAL
AT THREE POWER PLANTS WITH ANGLED SCREEN INTAKES

John A. Matousek
Robert W. Davis
Stephen J. Edwards*

Lawler, Matusky & Skelly Engineers
One Blue Hill Plaza, Pearl River, NY 10965, USA

ABSTRACT

Biological studies to evaluate angled screen intake systems were conducted at Niagara Mohawk Power Corporation's Oswego Steam Station Unit 6, a full-scale demonstration facility at Central Hudson Gas & Electric Corporation's Danskammer Point Generating Station, and New England Power Company's Brayton Point Generating Station Unit 4. These study sites are situated on three different water-body types - lake, river/estuary, and marine bay, respectively - permitting the evaluation of a large number of aquatic populations.

The three studies were designed to determine the effectiveness of the angled screens at diverting fish entrained with condenser cooling water from the intake as well as initial and extended survival of the fish collected from the diversion flow. Experimental variables incorporated into the study design at one or more of the test sites included bypass velocity, ratio of bypass velocity to angled screen approach velocity, screen mesh size, residency and predation, recirculation, photoperiod, and angled screen wash cycle. Diversion efficiency ranged from 76% at Brayton Point to 99% at Danskammer Point, with seasonal, species, and age (length) variability noted. With the exception of bay anchovy at Brayton Point, initial survival was high for all species. Extended survival evaluated for 96 h at Oswego and Danskammer Point and for 48 h at Brayton Point varied considerably. In general, two groups were identified at each site: a sensitive group composed of a few numerically dominant species (mostly Clupeiformes) that exhibited high postdiversion mortality and a hardy group (mixed species) with low postdiversion mortality.

The three studies indicate that an angled screen intake system is a successful physical barrier for mitigating fish impingement, has a high guiding capability, and demonstrates high initial survival following passage through the diversion system. Extended survival varied, but overall there was reasonably good survival for all but a few sensitive species.

*Present address: Environmental Services
80 Sunny Ridge Road, Easton, CT 06612, USA

INTRODUCTION

The amount of water used by industry each year, especially the non-consumptive use by electric utilities for condenser cooling, represents a significant portion of the total fresh water available. Initially, problems associated with this kind of water use concerned thermal discharges and their potential to alter the near-field temperature cycle. Attention then shifted to the cooling water intakes and their potential for entraining small planktonic organisms and entrapping larger organisms, with subsequent impingement on the intake traveling screens.

At the same time, greater public awareness of the need to protect the environment resulted in the passage and implementation of the Federal Water Pollution Control Act of 1972 (Public Law 92-500) and the Clean Water Act amendments of 1977 (PL 95-217). This legislation features sections pertaining to cooling water intakes, notably Section 316(b) of PL 92-500, which requires that the location, design, construction, and capacity of cooling water intake structures reflect the best technology available for minimizing adverse environmental impacts.

Intake design, including the incorporation of fish protection devices, is one way to minimize cooling water intake impact, thus achieving the goals of Section 316(b) (Hanson et al., 1977). Such protective devices fall into three general categories: collection and removal, deterrence, and diversion (Mussalli et al., 1978; Cannon et al., 1979).

Diversion devices depend on behavioral characteristics, and rely on the ability of fish to perceive and react to an external stimulus. Angled screens and louvers are the two primary diversion systems that have been evaluated for impact mitigation. Early studies on fish diversion systems were conducted in California and the Pacific

Northwest. The initial studies were concerned primarily with downstream passage of juvenile salmonids at hydroelectric facilities, and fish exclusion at water diversion projects. Both louver arrays and angled screens were employed, with good results reported for both systems.

Initial evaluation studies on the use of diversion systems at pumped cooling water intakes closely followed the studies conducted at hydroelectric facilities. Diversion transport systems were first evaluated in large-scale test flumes during the early 1970s (Schuler, 1974; Schuler and Larson, 1975; Taft et al., 1976; Taft and Mussalli, 1978; Taft et al., 1981). Results from these tests led to the construction of a full-scale angled screen intake system that permitted continued evaluation under actual environmental conditions and with naturally occurring fish populations. The Empire State Electric Energy Research Corporation (ESEERCO), a research group that comprises the New York State electric utilities that sponsored much of the large-scale laboratory studies on angled screens, sponsored the full-scale angled screen study at Central Hudson Gas & Electric Corporation's (CHGE) Danskammer Point Generating Station on the Hudson River. At approximately the same time, Niagara Mohawk Power Corporation incorporated an angled screen intake system at their Oswego Steam Station Unit 6 on Lake Ontario. The third angled screen intake system was constructed at New England Power Company's Brayton Point Station Unit 4 on Mount Hope Bay, Massachusetts, the northeast portion of Narragansett Bay.

Lawler, Matusky & Skelly Engineers (LMS) conducted multiyear biological evaluations at the three angled screen intake structures. Each intake system was evaluated to determine diversion efficiency, defined as the number of fish in the diversion system divided by the number of fish entering the intake, and survival of those fish in the diversion system. The diversion efficiency and survival information was used to calculate total system efficiency. Several

secondary parameters were incorporated in each study to obtain additional understanding of system operational conditions.

DESCRIPTION OF ANGLED SCREEN INTAKES

Danskammer Point Angled Screen Demonstration Facility

The Danskammer Point angled screen demonstration facility (Fig. 1) was located in the cooling water intake canal and consisted of two 3.0 m wide vertical traveling screens set at a 25° angle to the approach flow. The angled screen approach channel, 3.2 m wide and approximately 13.5 m long, led to a 3.0 m high by 15.2 cm wide bypass. The bypass contracted to a 50.0 cm diameter pipe and then to two 30.5 cm diameter pipes that conveyed the water to two shrouded Hidrosta (Model L12F) screw-impeller centrifugal pumps. Cooling water approach velocity to the angled screens was adjustable, resulting in test velocities of 15.2, 30.5, and 45.7 cm/s. Water depth in the angled screen approach channel was tidally influenced and ranged from 2.0 to 3.0 m. The screens were designed to allow the interchange of standard (9.5 mm) and fine (1.0 mm) mesh screening panels.

Oswego Steam Station Unit 6

The Oswego Steam Station Unit 6 cooling water is withdrawn from Lake Ontario via a submerged inlet with a velocity cap, circulated through the condensers, and returned to the lake through a submerged jet diffuser. The circulating water flow enters the intake screenhouse through a vertical intake shaft, passes through trash racks with 7.6 cm openings, and enters two screenbays in the primary screenwell, each 5.2 m wide, with a water column depth that varies from 7.3 to 10.1 m depending upon pump operation and lake level (Fig. 2).

FIGURE 1

DANSKAMMER POINT ANGLED SCREEN DEMONSTRATION FACILITY

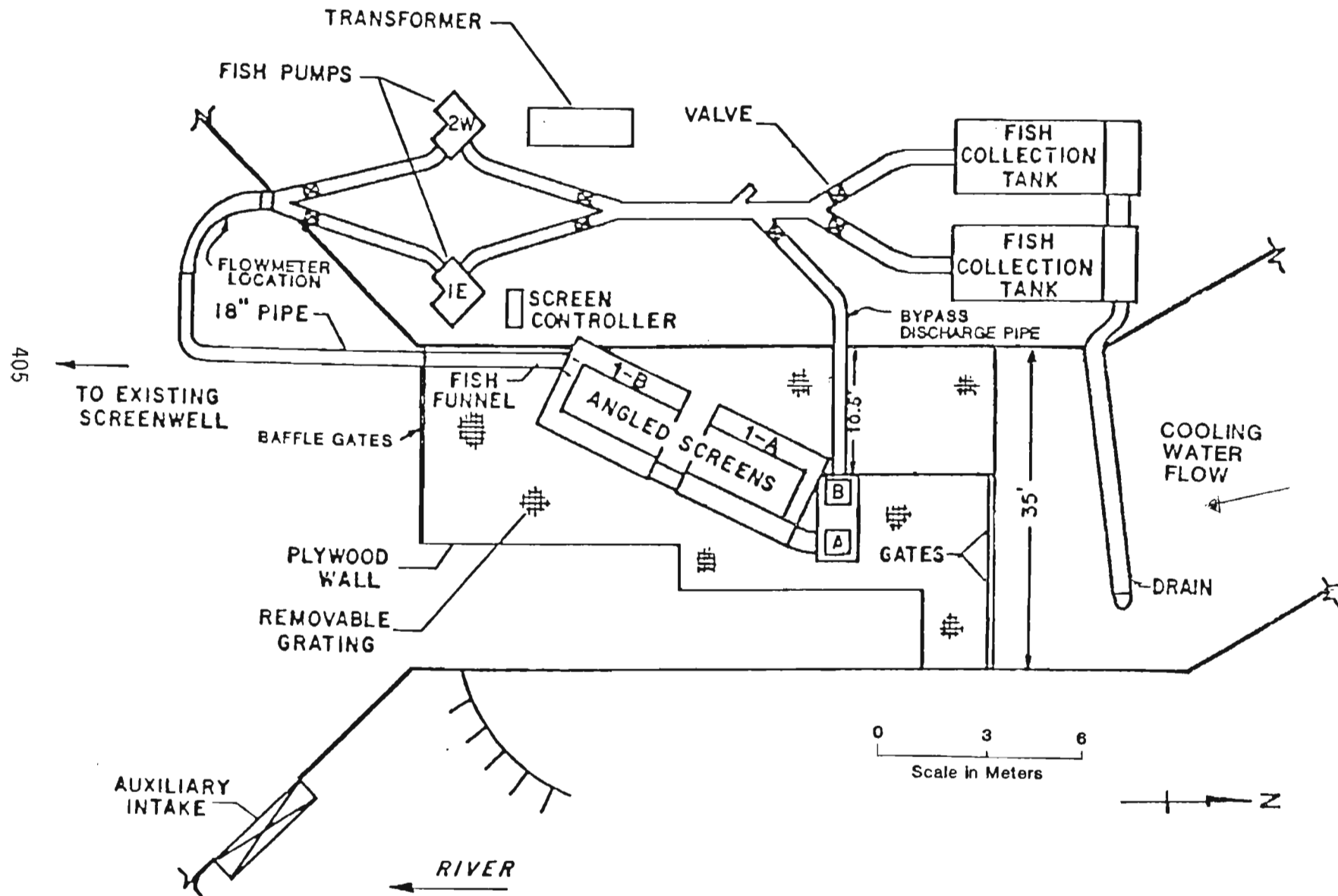
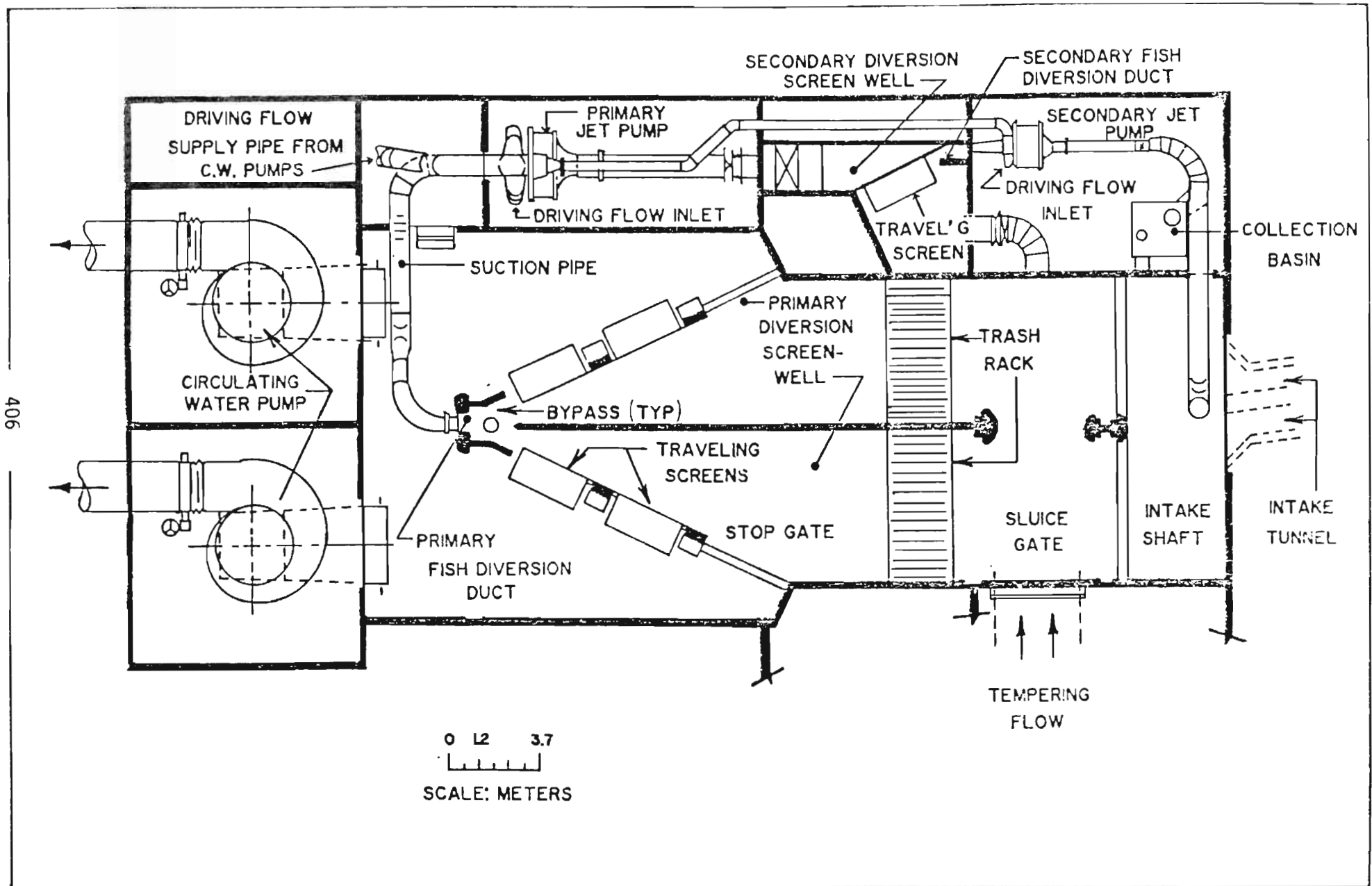


FIGURE 2
 OSWEGO STEAM STATION UNIT 6
 ANGLED SCREEN INTAKE



Each bay will accommodate three 3.0 m wide flush-mounted traveling screens separated by 1.0 m wide concrete piers. At present, each bay has two screens leading to a 15.2 cm wide bypass. Each bypass slot extends the full depth of the water column. The screens are angled 25° to the direction of flow, with their downstream ends converging but separated by a 1.5 m wide wall.

The bypass suction flow induced by an 82.3 cm diameter jet pump is designed such that the ratio of the average angled screen approach velocity to the average bypass entrance velocity is 1:1. The two bypass slots converge in the horizontal plane as well as the vertical plane at a 45° angle to two 61.0 cm diameter pipes. The two pipes join into a single 82.3 cm diameter pipe that becomes the suction side of the primary peripheral jet pump. The primary jet pump discharges to a 1.6 m wide secondary screenwell.

The secondary screenwell contains one angled traveling screen identical in design and orientation to the main screens except for its depth. The water depth in the secondary bay varies from 2.4 to 4.6 m, depending on lake elevation and the number of cooling water pumps operating. Most of the water discharged from the primary jet pump flows through the secondary screen and is returned to the primary screenwell through a 1.1 m diameter pipe. The fish move across the secondary screen into another 15.2 cm wide bypass slot that extends the full depth of the water column. The secondary bypass slot converges in the vertical plane to a 45.7 cm diameter pipe. At the secondary jet pump this pipe is reduced to a 42.7 cm diameter suction pipe. The ratio of the average secondary bay angled screen approach velocity to the average secondary bypass velocity varies from 1:1 to 1:1.3. The secondary jet pump discharges into a 76.2 cm diameter discharge pipe embedded in the roof of the intake tunnel. The pipe extends approximately 280.0 m offshore where it rises vertically and terminates as a horizontal discharge approximately 2.0 m off the bottom and 83.0 m from the intake.

Brayton Point Generating Station Unit 4

Cooling water enters the Brayton Point Unit 4 intake structure through eight 3.3 m wide by 4.2 m high openings (Fig. 3). The openings are shielded by trash racks with a bar spacing of 7.6 cm on center. Ten meters into the mouth of the screenwell, the width constricts to 12.3 m approaching a 1.5 m thick center wall that divides the structure in half. Each half is equipped with three 3.0 m wide flush-mounted vertical traveling screens set at 25° to the approaching flow. Each screen panel is modified with a fish bucket to maintain the impinged fish in water as the screen rotates. Impinged fish are washed off the back of the screens by a low-pressure spray into a fish return trough that discharges into the Lee River. The screens are designed to allow the interchange of standard (9.5 mm) and fine (1.0 mm) mesh screening panels. Cooling water exits the angled screen structure into a cooling canal.

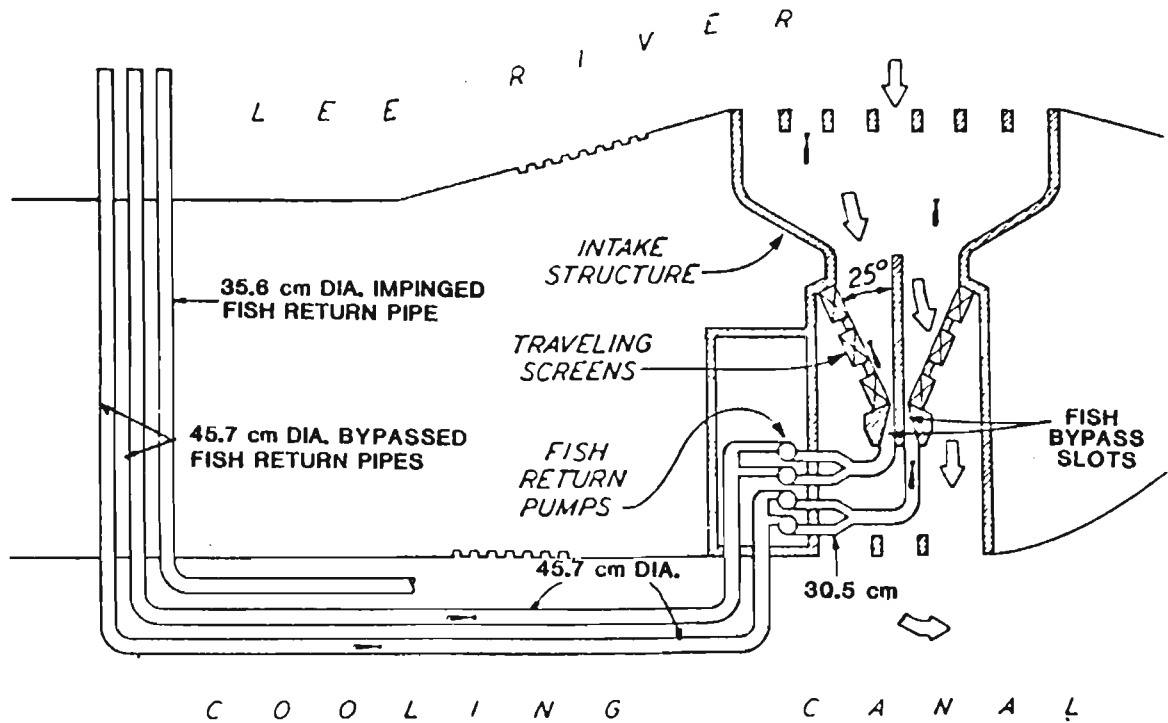
The design flow of 17.2 m³/s is drawn through the bar racks at an average approach velocity of 15.2 cm/s. As the structure constricts, the design velocity increases to 30.5 cm/s approaching the traveling screens. Approximately 97% of the design flow (16.75 m³/s) is drawn through the screens; the remaining 3% (0.5 m³/s) is pumped through the fish bypass.

At the apex of each screenwell a fish bypass is provided (Fig. 3). Each bypass has a rectangular opening 15.2 cm wide by 5.1 m high that constricts to a 45.7 cm diameter bypass pipe. The bypass slot is sized such that two shrouded 30.5 cm diameter screw impeller centrifugal pumps (Hidrostaal Model L12F) can induce a velocity of 30.5 cm/s at the slot's entrance. The bypass velocity at the slot is scheduled to be the same as the screen approach velocity.

FIGURE 3

BRAYTON POINT GENERATING STATION UNIT 4

ANGLED SCREEN INTAKE



SCALE: None

Each bypass opening leads to a 45.7 cm diameter fiberglass pipe that branches into two 30.5 cm diameter pipes that pass through one of two Hidrostat pumps, then back to a 45.7 cm diameter pipe that carries the flow to the Lee River approximately 91 m downstream of the intake structure.

METHODS

At Danskammer Point juvenile and older fish were collected on a seasonal basis from the fish pump discharge using nets (abundance) or collection tanks from which the fish could be removed and evaluated for extended survival (96 h). Nets located on the screenwash discharge were used to collect all fish impinged on the angled screens.

At Oswego Steam Station Unit 6 the discharge flow was diverted into a 2.4 by 2.4 m collection basin during biological performance sampling. Nets located on the primary and secondary screenwash discharge were used to collect all fish impinged on the angled screens.

At Brayton Point Station Unit 4 impingement abundance and survival collections were conducted simultaneously with bypass abundance and survival collections. The former collections were made from two built-in fiberglass collection tanks. The latter collections were made from two collection nets attached to a sampling port on each 45.7 cm bypass return line.

Small buckets, nylon nets, and dip nets were used at all three study sites to transfer fish to be evaluated for extended survival from the collection devices to flow-through extended survival holding containers. Juvenile or young-of-the-year fish were held primarily in 19 L plastic buckets. Larger fish or large numbers of individual species were held in 568 L linear polyethylene holding

tanks fitted with a central standpipe arrangement that permitted a uniform exchange of water throughout the water column. Water was supplied to the bucket and tank holding containers through a header system and regulated by valves to maintain ambient conditions for the extended survival period.

Angled screen diversion efficiency was calculated using the following formula:

$$\text{Diversion Efficiency (DE)} = \frac{D}{D+I} \times 100$$

where:

D = number of fish collected from the diversion flow
I = number of fish collected from the angled screens

Initial survival was determined by observing fish from the diversion flow immediately following collection. Fish were identified and classified (live, stunned, dead), with the classification conditions defined as follows:

Live - swimming normally, no orientation problem

Stunned - swimming erratically, swimming on their side, struggling

Dead - no vital life signs, no body or opercular movement, no response to gentle probing

Initial survival was calculated by dividing the number of fish determined to be initially live (L_I) by the number initially determined to be live, stunned (ST_I), and dead (D_I). The initial survival formula is:

$$\text{Initial Survival (S}_I\text{)} = \frac{L_I}{L_I + ST_I + D_I}$$

Extended survival (S_E) observations were conducted over 96 h for the Danskammer Point and Oswego Steam Station Unit 6 evaluation programs and for 48 h for the Brayton Point Unit 4 program. Extended survival was calculated by dividing the number of fish classified live at the end of the observation period by the total number of fish examined for extended survival.

Angled screen system efficiency is basically a determination of the probability that a fish entering the angled screen system will survive some period of time after entry and diversion.

The diversion system efficiency (SE) is determined by:

$$SE = DE \times S_I \times S_E$$

This formula also includes all types of incidental mortality caused by collection, handling, and other nondiversion-related stress.

Impingement survival was determined for fish collected from the modified angled traveling screens at Brayton Point Unit 4. Total system efficiency (TSE) was obtained by correcting impingement efficiency ($IE = 1 - DE$) and diversion efficiency (DE) for initial and extended survival:

$$TSE = (IE \times S_I \times S_E) + (DE \times S_I \times S_E)$$

RESULTS AND DISCUSSION

The most numerous species collected at each plant and their occurrence by water body is presented in Table 1.

Table 1. Species inventory

Common name	Scientific name ^a	Hudson River	Lake Ontario	Mount Hope Bay
Alewife	<u>Alosa pseudoharengus</u>	X	X	X
American eel	<u>Anguilla rostrata</u>	X	X	X
American shad	<u>Alosa sapidissima</u>	X		
Atlantic menhaden	<u>Brevoortia tyrannus</u>	X		X
Atlantic silverside	<u>Menidia menidia</u>			X
Atlantic tomcod	<u>Microgadus tomcod</u>	X		X
Bay anchovy	<u>Anchoa mitchilli</u>	X		X
Blueback herring	<u>Alosa aestivalis</u>	X		
Brown bullhead	<u>Ictalurus nebulosus</u>	X		
Butterfish	<u>Peprilus triacanthus</u>			X
Emerald shiner	<u>Notropis atherinoides</u>		X	
Fourspine stickleback	<u>Apeltes quadracus</u>			X
Gizzard shad	<u>Dorosoma cepedianum</u>	X	X	
Hogchoker	<u>Trinectes maculatus</u>	X		X
Mottled sculpin	<u>Cottus bairdi</u>		X	
Northern pipefish	<u>Syngnathus fuscus</u>			X
Pumpkinseed	<u>Lepomis gibbosus</u>	X	X	
Rainbow smelt	<u>Osmerus mordax</u>	X	X	X
Seaboard goby	<u>Gobiosoma ginsburgi</u>			X
Spottail shiner	<u>Notropis hudsonius</u>	X	X	
Striped bass	<u>Morone saxatilis</u>	X		
Tautog	<u>Tautoga onitis</u>			X
Threespine stickleback	<u>Gasterosteus aculeatus</u>		X	X
White catfish	<u>Ictalurus catus</u>	X		
White perch	<u>Morone americana</u>	X	X	X
Winter flounder	<u>Pseudopleuronectes americanus</u>			X

^aAmerican Fisheries Society (1980)

Danskammer Point Angled Screen Demonstration Facility

Spring, fall, and winter (limited) sampling, periods of normally high fish abundance, resulted in the collection of 59,309 fish consisting of 38 species and 21 families (Table 2) (ESEERCO, 1985). Diversion efficiency was high for all species, ranging from 95.4 to 100.0% and was 99.4% for all species combined. Diversion efficiency was similar among seasonal sampling periods and angled screen approach velocities. No influence on diversion efficiency was determined for screen mesh size, time of day, or angled screen approach velocity.

Initial survival (S_I) observations of 16,555 fish collected from the diversion flow (Table 1) ranged from 80.9% for the dominant bay anchovy to 100.0% for the hogchoker, and was 90.2% for all species combined. No seasonal or angled screen approach velocity effect was noted on initial fish survival. Extended 96 h survival observations were made on 13,007 fish (Table 1) and resulted in a survival value (S_E) of 35.3% for all fish combined. In general, the fish were divided into two groups: a sensitive group comprising bay anchovy and three anadromous herrings (alewife, American shad, and blueback herring) and a hardy group containing the majority of fish species tested, including white perch, spottail shiner, pumpkinseed, Atlantic tomcod, and striped bass.

The system efficiency calculated for the demonstration facility was 31.6% for total fish. System efficiency values for the major species exhibited an extremely wide range, but generally fell into two groups: a group with low SE (due primarily to low extended survival) that included bay anchovy, which was extremely sensitive, and herrings, and a group with SE values greater than 70.0%, characterized by white perch and spottail shiner.

Table 2. Fish collection information, diversion efficiency, initial and extended survival, and system efficiency: Danskammer Point angled screen demonstration facility, 1981-1983

Species	Number collected	Percent composition	Diversion efficiency (%)	Initial survival		Extended survival		System efficiency SE (%)
				Number analyzed	S _I (%)	Number analyzed	S _E (%)	
Bay anchovy	26,948	45.4	99.6	6,019	80.9	3,869	0.1	0.1
Blueback herring	11,331	19.1	99.6	4,166	94.9	3,683	8.2	7.8
White perch	10,415	17.6	99.5	3,521	95.3	3,217	84.6	80.2
Spottail shiner	2,331	3.9	99.3	789	99.4	569	86.8	85.7
Alewife	1,443	2.4	99.0	390	90.3	350	17.4	15.6
Pumpkinseed	1,067	1.8	98.8	214	96.7	210	92.4	88.3
Atlantic tomcod	898	1.5	97.9	211	96.2	208	77.4	72.9
White catfish	693	1.2	99.7	243	99.2	208	94.7	93.7
Brown bullhead	594	1.0	99.8	213	98.6	187	88.2	86.8
American shad	520	0.9	99.4	164	96.3	147	8.8	8.4
Hogchoker	461	0.8	96.3	133	100.0	55	98.2	94.6
Striped bass	426	0.7	98.4	133	90.2	131	67.2	59.6
Others (28 taxa)	2,182	3.7	98.7	359	95.0	173	81.5	76.4
Total fish	59,309		99.4	16,555	90.2	13,007	35.3	31.6

Following summarization of the ESEERCO-sponsored study results, a second angled screen study sponsored by CHGE was conducted at the Danskammer Point demonstration facility. The follow-up study concentrated on diversion efficiencies where the bypass velocity was not equal to the angled screen approach velocity (LMS, 1986). The study consisted of 12 weeks of sampling during fall 1984, a period characterized by fish populations composed primarily of juveniles, and five weeks during spring 1985, when juvenile Atlantic tomcod and spawning populations of alewife, blueback herring, and white perch predominated. The collection procedure was modified to incorporate a crowder, which concentrated the diverted organisms in trays from which they could be distributed into the long-term holding containers. Collection and sorting were thus carried out with minimal organism contact.

Four angled screen approach and bypass velocities were evaluated: 15.2, 30.5, 45.7, and 61.0 cm/s. The study design called for continuous sampling between 2100 and 0900 h (period of greatest fish abundance) under randomly selected angled screen approach and bypass velocity combinations. Under all tests the bypass velocity, calculated at the entrance to the diversion system, would equal or be less than the angled screen approach velocity; bypass velocity would never exceed the approach velocity. The program design resulted in 10 different velocity combinations and six separate angled screen approach to bypass velocity ratios.

During the two seasonal sampling periods 22,125 fish representing 35 species and 18 families were collected. Abundance, percent composition, diversion efficiency, survival information, and system efficiency for the major species and total fish collected over the fall and spring sampling periods are presented in Table 3. As noted for the three-year ESEERCO-sponsored study, diversion efficiency was high for all species, ranging from 93.0 to 100.0%, with an overall value of 98.7% for all species combined. No difference in

Table 3. Fish collection information, diversion efficiency, initial and extended survival, and system efficiency: Danskammer Point angled screen demonstration facility, fall 1984–spring 1985

Species	Number collected	Percent composition	Diversion efficiency (%)	Initial survival		Extended survival		System efficiency SE (%)
				Number analyzed	S_I (%)	Number analyzed	S_E (%)	
Blueback herring	6,176	27.9	98.9	4,793	98.0	4,682	52.9	51.3
Bay anchovy	5,413	24.5	98.9	4,157	82.2	3,289	15.2	12.4
White perch	4,432	20.0	99.7	3,110	97.3	3,064	94.9	92.1
Atlantic tomcod	1,784	8.1	99.6	1,139	97.1	1,116	87.9	85.0
Hogchoker	1,538	7.0	93.0	1,343	99.9	1,322	99.1	92.1
White catfish	574	2.6	99.5	415	99.5	276	98.2	97.2
Spottail shiner	485	2.2	99.8	331	99.7	330	98.8	98.3
Alewife	261	1.2	97.7	187	94.1	183	38.8	35.7
Pumpkinseed	212	1.0	99.1	159	98.1	159	96.9	94.2
Rainbow smelt	159	0.7	100.0	122	91.8	108	51.9	47.6
Striped bass	146	0.7	100.0	101	96.0	98	93.9	90.1
American shad	120	0.5	98.3	90	97.8	89	61.8	59.4
Others (23 taxa)	815	3.7	99.1	620	96.5	519	63.6	60.8
Total fish	22,125		98.7	16,567	93.9	15,235	62.6	58.0

DE was determined for the six different ratios of angled screen approach to bypass velocity.

A total of 16,567 fish collected from the diversion flow were evaluated for initial survival. With the exception of bay anchovy (S_I 82.2%), initial survival for the major species was greater than 90%, and was 93.9% for all fish combined.

Extended survival observations were conducted for 96 h following collection. A total of 15,235 fish were observed for extended survival (S_E); 14,857 were classified as live and 378 as stunned. At the end of the observation period 9531 fish were classified as live and 34 as stunned. Extended survival for the major species was variable, ranging from 15.2% for bay anchovy to 99.1% for hogchoker. Overall, the S_E was 62.6%, 27.3 percentage points higher than the ESEERCO S_E of 35.3%, which is directly related to the modified collection method.

As noted for the three-year angled screen study conducted at the Danskammer Point demonstration facility between February 1981 and October 1983, fish populations are generally grouped in one of two categories: a sensitive group and a hardy group. The sensitive group, characterized by high mortality over the 96 h observation period, included the three Alosa spp. (alewife, American shad, and blueback herring), bay anchovy, and rainbow smelt; the hardy group exhibited low mortality over the 96 h observation period and included the two temperate basses (striped bass and white perch), sunfish (represented by the pumpkinseed), catfish (represented by the white catfish), spottail shiner, Atlantic tomcod, and hogchoker. The 1984-1985 system efficiency (SE) of 58.0% is almost twice the SE value obtained from the 1981-1983 ESEERCO-sponsored study of 31.6%.

Oswego Steam Station Unit 6

Two species (Table 4), alewife and rainbow smelt, accounted for approximately 90% of fish entrapped at the Oswego Steam Station Unit 6 intake during the April 1981 - March 1983 study period (LMS, 1984). Diversion efficiency was 79.3% for alewife and 74.2% for rainbow smelt. The combined DE was 77.9%, ranging from 53.4% for mottled sculpin to 94.8% for gizzard shad.

For the seven most numerous species a total of 34,294 individuals were examined for initial survival. The combined S_I was 50.9% and ranged from a low of 45.2% for rainbow smelt to 87.4% for emerald shiner. Extended survival observations covering 96 h were conducted on 7534 fish. Alewife exhibited the lowest S_E , at 22.4%; the highest, 93.6%, was calculated for mottled sculpin. The combined S_E for the top seven species was 35.4%.

System efficiency (SE) calculated for the dominant species represents fish diversion across two separate angled screen systems and passage through two jet pumps on the bypass return system. The calculated SE was 14.0%, and ranged from 9.2% for alewife to 72.7% for spottail shiner.

Brayton Point Generating Station Unit 4

During the 18 month (October 1984 - March 1986) study period, 79,206 fish were entrapped (based on corresponding samples between angled screens and diversion flow): 18,791 collected from the angled screens and 60,415 estimated from the diversion flow (LMS, 1987). The angled screen evaluation program resulted in the collection of 57 fish species representing 36 families. Abundance, percent composition, diversion efficiency, survival information, and system efficiency for the major species (95.4% of the total) and total fish collected during the 18 month study period are pre-

Table 4. Fish collection information, diversion efficiency, initial and extended survival, and system efficiency: Oswego Steam Station Unit 6, April 1981-March 1983

Species	Number ^a collected	Percent composition	Diversion efficiency (%)	Initial survival		Extended survival		System efficiency SE (%)
				<u>S_I</u> Number analyzed	<u>S_I</u> S _I (%)	<u>48 h S_E</u> Number analyzed	<u>48 h S_E</u> S _E (%)	
Alewife	448,870	46.0	79.3	12,496	51.8	3,599	22.4	9.2
Rainbow smelt	433,862	44.5	74.2	18,215	45.2	2,846	36.4	12.2
Emerald shiner	22,598	2.3	94.2	812	87.4	367	87.5	72.0
Gizzard shad	26,173	2.7	94.8	1,393	73.7	249	47.8	33.4
White perch	18,808	1.9	90.6	475	55.6	229	69.4	35.0
Mottled sculpin	11,827	1.2	53.4	460	81.5	125	93.6	40.7
Spottail shiner	12,744	1.3	92.9	443	86.2	119	90.8	72.7
Total	974,882		77.9	34,294	50.9	7,534	35.4	14.0

^aEstimated abundance based on average monthly collection rates (No./h) over the period April 1981 - March 1983.

sented in Table 5. Diversion efficiency (DE) was 76.3% for all species combined. Over 93% of the total bay anchovy entrapment occurred during August and September 1985 when fine mesh screen panels were being tested. The bay anchovy collected during this period were early juveniles that exhibited very low diversion. Diversion efficiency ranged from 52.9% for northern pipefish to 99.4% for American eel. Diversion efficiency was 86.7% for the other 45 fish species combined.

Initial survival (S_I) observations of 28,186 fish collected from the diversion flow ranged from 5.6% for bay anchovy to 99.7% for American eel. S_I was 57.8% for all fish combined, increasing to 82.6% with the exclusion of bay anchovy. Extended survival observations (S_E) were made on 9209 fish and resulted in an overall survival value (S_E) of 63.4%. Extended survival trends for the major species were similar to initial survival and were variable among species, ranging from 0% for bay anchovy to 99.6% for tautog. In general, the fish were divided into two groups: a sensitive group comprising bay anchovy, Atlantic silverside, Atlantic menhaden, and butterfish and a hardy group containing a majority of the fish species tested, including winter flounder, northern pipefish, threespine stickleback, fourspine stickleback, tautog, American eel, hogchoker, and seaboard goby.

Angled screen system efficiency (SE) for all fish combined was 28.0% and increased to 47.7% with the exclusion of the bay anchovy. Survival efficiency values for the less tolerant group ranged from 0.0% for bay anchovy to 22.5% for Atlantic silverside. SE values for the hardy species ranged from a low of 55.2% for seaboard goby to 90.0% for the threespine stickleback.

The Brayton Point Unit 4 traveling screens were modified with lip troughs and a low-pressure rinse to increase survival of impinged fish. Initial survival (S_I) observations of 18,831 fish col-

Table 5. Fish collection information, diversion efficiency, initial and extended survival, and system efficiency: Brayton Point Station Unit 4, October 1984–March 1986

Species	Number collected	Percent composition	Diversion efficiency (%)	Initial survival		Extended survival		System efficiency SE (%)
				Number analyzed	S_I (%)	Number analyzed	S_E (%)	
Bay anchovy	32,563	41.1	57.0	9,095	5.6	144	0.0	0.0
Atlantic silverside	23,504	29.7	96.9	10,338	81.8	2,742	28.4	22.5
Winter flounder	8,284	10.4	87.7	3,114	96.1	2,409	91.6	77.2
Northern pipefish	3,284	4.1	52.9	837	96.2	799	71.2	36.2
Threespine stickleback	1,481	1.9	92.4	621	98.6	600	98.8	90.0
Atlantic menhaden	1,279	1.6	90.3	606	16.0	97	17.5	2.5
Fourspine stickleback	1,108	1.4	83.5	448	98.4	439	91.6	75.3
Tautog	843	1.1	62.2	256	99.2	253	99.6	61.5
American eel	837	1.1	99.4	391	99.7	27	85.2	84.4
Butterfish	819	1.0	95.5	381	49.9	189	13.2	6.3
Hogchoker	811	1.0	85.6	326	99.4	322	96.6	82.2
Seaboard goby	750	0.9	83.2	294	91.2	265	72.8	55.2
Others (45 taxa)	3,643	4.6	86.7	1,479	64.2	923	50.5	28.1
Total fish	79,206		76.3	28,186	57.8	9,209	63.4	28.0
Total excluding bay anchovy	46,643		89.7	19,091	82.6	9,065	64.4	47.7

lected from the angled vertical traveling screens (Table 6) ranged from 1.7% for the dominant bay anchovy to 99.1% for hogchoker. S_I was 24.3% for all fish combined, increasing to 89.6% with the exclusion of bay anchovy. Extended 48-h survival observations were made on 3855 impinged fish and resulted in an overall survival value (S_E) of 77.7%. Extended survival trends for the major taxa were similar to initial survival and were variable among species, ranging from 1.7% for bay anchovy to 98.4% for tautog.

The total system efficiency (TSE) was 32.4% (Table 6). TSE increased to 55.3% with the exclusion of bay anchovy, which constituted 41.1% of the total entrapment. TSE values for the major species exhibited an extremely wide range, from less than 0.1% for bay anchovy to 97.9% for tautog, but generally fell into two groups: a hardy group with TSE values greater than 65% and a sensitive group with TSE values less than 25%.

The primary contributors to lower total system efficiency were low initial (S_I) survival and/or extended (S_E) survival for the sensitive species. Atlantic silverside (29.7% of the total with 96.9% DE), a representative of the sensitive group with a TSE of 23.1%, exhibited high initial survival from both impingement (82.1%) and diversion (81.8%), but low extended survival, 22.2 and 28.4%, respectively.

DISCUSSION OF ANGLED SCREEN EVALUATION RESULTS

Angled screen diversion efficiency (DE) was similar at Oswego Steam Station Unit 6 (77.9%) and Brayton Point Station Unit 4 (76.3%), averaging approximately 77.0%. The Danskammer Point angled screen demonstration facility had an overall DE of 99.3%.

Based on results from tagging studies, diversion efficiency at the Oswego Steam Station was found to be affected by residency in the

Table 6. Initial and extended survival information on major taxa and total fish and calculated total system efficiency: Brayton Point Generating Station Unit 4, October 1984 - March 1986

Species	Impingement					Diversion					Total system efficiency TSE (%)
	Impingement efficiency (%)	Initial survival		Extended survival		Diversion efficiency (%)	Initial survival		Extended survival		
		S _I		48 h	S _E		S _I		48 h	S _E	
		Number observed	S _I (%)	Number observed	S _E (%)		Number observed	S _I (%)	Number observed	S _E (%)	
Bay anchovy	43.0	13,987	1.7	235	1.7	57.0	9,095	5.6	144	0	<0.1
Atlantic silverside	3.1	745	82.1	491	22.2	96.9	10,338	81.8	2,742	28.4	23.1
Winter flounder	12.3	1,025	95.6	787	95.2	87.7	3,114	96.1	2,409	91.6	88.4
Northern pipefish	47.1	1,551	98.1	1,134	95.1	52.9	837	96.2	799	71.2	80.2
Threespine stickleback	7.6	113	93.8	105	96.2	92.4	621	98.6	600	98.8	96.9
Atlantic menhaden	9.7	126	38.1	48	8.3	90.3	606	16.0	97	17.5	2.8
Fourspine stickleback	16.5	183	86.9	155	96.1	83.5	448	98.4	439	91.6	89.0
Tautog	37.8	329	97.9	317	98.4	62.2	256	99.2	253	99.6	97.9
American eel	0.6	5	60.0	-	-	99.4	391	99.7	27	85.2	84.4
Butterfish	4.5	37	56.8	21	57.1	95.5	381	49.9	189	13.2	7.7
Hogchoker	14.4	117	99.1	115	96.5	85.6	326	99.4	322	96.6	96.0
Seaboard goby	16.8	126	87.3	109	85.3	83.2	294	91.2	265	72.8	67.7
Others (45 taxa)	13.3	487	70.2	338	80.8	86.7	1,479	64.2	923	50.5	35.7
Total	23.7	18,831	24.3	3,855	77.7	76.3	28,186	57.8	9,209	63.4	32.4
Total excluding bay anchovy	10.3	4,844	89.6	3,620	82.6	89.7	19,091	82.6	9,065	64.4	55.3

intake screenwell. During construction of the Unit 6 intake, several 20.0 cm pipes were placed in the intake screenwell and extended vertically along the center wall from above the water surface to the bottom of the screenwell and across the screenwell floor. A 1.5 m high plate at the bottom of each traveling screen precludes water withdrawal from the bottom of the screenwell, and corrugated stop logs at the location for the addition of the third traveling screen in each screenwell offers fish quiescent or refuge areas.

The Danskammer Point angled screen demonstration facility was designed to permit evaluation studies at varying angled screen and diversion bypass velocities. The primary sampling program incorporated angled screen approach velocities ranging from 15.2 to 61.0 cm/s, with the velocity at the diversion bypass opening matched to the angled screen approach velocity. Diversion efficiency was greater than 99.0% for all velocities tested and there was no relationship determined between velocity and diversion efficiency. Various ratios of angled screen approach and bypass velocities were tested under the CHGE-sponsored program. No difference in angled screen diversion efficiency was noted for the various angled screen approach and bypass velocity combinations although there was a general trend of increased DE with increased bypass velocity.

The Danskammer Point angled screens were designed to permit the installation of either 9.5 or 1.0 mm mesh panels. As operational problems with the year-round use of the 1.0 mm mesh panels were found to be negligible, the second half of the ESEERCO study for juvenile and adult fish was conducted using fine mesh screens. Diversion efficiency was similar among comparable seasons with 9.5 and 1.0 mm mesh screens. Detailed analysis was done using the dominant bay anchovy, and no relationship between panel mesh size and diversion efficiency was found. Evaluation of the Brayton Point collection information from 9.5 and 1.0 mm mesh screens indicates that diversion is directly related to the size (age) of the fish,

with small fish exhibiting greater impingement and larger, more mature fish with greater sensory and swimming capabilities exhibiting greater diversion (LMS, 1987).

Overall, the angled screen intake systems were found to be effective at diverting fish from the intake screenwell to the bypass return system. The degree of effectiveness varied by species, seasonal condition of the population, e.g., prespawn and postspawn stocks, and the age of the fish.

The critical factor in assessing the effectiveness of a mitigative intake system is the condition of the fish diverted or returned to the source water body. Initial survival at the Danskammer Point demonstration facility was 90.2% for the ESEERCO study and 93.9% for the CHGE study. Initial survival was 50.9% at Oswego Steam Station Unit 6 and 57.8% at Brayton Point Unit 4. Extended survival at Danskammer Point was 35.3% for the ESEERCO-sponsored study and 62.6% for the CHGE-sponsored study. Extended survival was 35.4% at Oswego Steam Station Unit 6 and 63.4% at Brayton Point Unit 4. At all three locations, collection and handling mortality has been determined to be or is projected to be significant. The initial work at Danskammer Point suggested that a change in the survival collection methodology, in which all netting was eliminated and fish were not directly handled, would greatly decrease stress and increase survival. As noted in the CHGE study, when the collection net was eliminated and handling minimized, initial survival increased 3.9% and extended survival increased 43.6%. The most dramatic changes were observed for extremely sensitive species, such as bay anchovy, with S_E increasing from 0.1 to 15.2%, and alewife, with the 96 h S_E increasing from 17.4 to 38.8% (Table 7). The Oswego Steam Station method of collection was to use a crowder to concentrate the fish in a collection basin and then dipnet them out to holding containers. At Brayton Point nets suspended in the water were used to intercept the bypass flow.

Table 7. Comparable information on common taxa and total fish among the angled screen intake systems evaluated

		DANSKAMMER POINT		OSWEGO STEAM	BRAYTON POINT
		ESEERCO	CHGE	STATION UNIT 6	STATION UNIT 4
Alewife	DE	99.0	97.7	79.3	93.0
	S _I	90.3	94.1	51.8	52.2
	S _E	17.4	38.8	22.4	12.5
	SE	15.6	35.7	9.2	6.0
Atlantic tomcod	DE	97.9	99.6	-	94.6
	S _I	96.2	97.1	-	84.1
	S _E	77.4	87.9	-	48.9
	SE	72.9	85.0	-	38.9
Bay anchovy	DE	99.6	98.9	-	57.1
	S _I	80.9	82.2	-	5.6
	S _E	0.1	15.2	-	0.0
	SE	<0.1	12.4	-	0.0
Rainbow smelt	DE	100.0	100.0	74.2	94.7
	S _I	76.9	91.8	45.2	11.9
	S _E	10.0	51.9	36.4	28.6
	SE	7.7	47.6	12.2	3.2
Spottail shiner	DE	99.3	99.8	92.9	-
	S _I	99.4	99.7	86.2	-
	S _E	86.8	98.8	90.8	-
	SE	85.7	98.3	72.7	-
White perch	DE	99.5	99.7	90.6	92.7
	S _I	95.3	97.3	55.6	92.0
	S _E	84.6	94.9	69.4	39.1
	SE	80.2	92.1	35.0	33.3
Total fish	DE	99.4	98.7	77.9	76.3
	S _I	90.2	93.9	50.9	57.8
	S _E	35.3	62.6	35.4	63.4
	SE	31.6	58.0	14.0	28.0

DE - Diversion efficiency
S_I - Initial survival
S_E - Extended survival
SE - System efficiency

Water depth at the collection site was minimal and turbulence in the collection net substantially increased the stress placed on the organisms collected for survival observations.

During the various Danskammer Point velocity tests, no relationship was observed for initial survival, but there was a general trend of greater survival at lower velocities for sensitive species and little or no influence on hardy species. Aside from this trend, survival was not influenced by varying the ratio of angled screen approach to bypass approach velocity.

In general, all three studies indicate that older fish of a species exhibit greater initial and extended survival. However, in all three studies the influence was species specific, with survival greater in the younger age groups for some taxa due to the condition of the post spawned adults.

The three angled screen systems having a high guiding capability, were found to be effective devices for mitigating fish impingement, and with the exception of a few species, demonstrated high initial survival of fish following passage through the diversion system. Extended survival varied by species and was greatly influenced by collection and handling techniques. Angled screen intake system efficiencies (all species combined) ranged from a low of 14.0% at Oswego Steam Station Unit 6 to a high of 58.0% at Danskammer Point under the CHGE-sponsored study. The angled screen intake system was very effective for the hardy species but relatively ineffective for the numerically dominant sensitive species. In the actual operation of an angled screen intake system the fish would not be collected and observed, but returned directly to the source water body. Thus, the true measure of system efficiency should not include mortality associated with collection and handling.

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POTENTIAL IMPACT OF TIDAL POWER DEVELOPMENT ON ESTUARINE
ENVIRONMENTS IN THE SOUTHEASTERN USA

Roger A. Rulifson
Institute for Coastal and Marine Resources
and
Department of Biology
East Carolina University
Greenville, NC 27858

Michael J. Dadswell
Department of Biology
Acadia University
Wolfville, Nova Scotia B0P 1X0 Canada

George C. Baker
Tidal Power Corporation
Halifax, Nova Scotia B3J 1P3 Canada

ABSTRACT

Harnessing the power of ocean tides for the production of electricity is now a reality. The world potential for tidal hydropower production is estimated at 1,240 billion KWh/yr, with at least 50 coastal areas around the world considered as potential sites for power development. As the technology of low-head turbines improves, the number of potential power sites will increase to include southeastern USA coastal areas that possess strong tidal currents. Countries including Canada, France, The People's Republic of China, and the Soviet Union have constructed commercial tidally-powered electrical generation plants, yet only Canada has conducted extensive studies to determine potential impact of tidal power on aquatic resources. Changes in hydrology, primary productivity, and secondary productivity associated with tidal power can be extensive depending on the design of the facility and mode of operation. Studies to determine effects of operational tidal power plants on fisheries are limited in number and scope. Damage is greatest in regions where fish are abundant and fish passage is repeated by the same population many times throughout the year. Three tidal power designs and their potential environmental effects are discussed relative to tidal power development in the southeastern USA.

INTRODUCTION

Tidal hydropower -- the production of electricity by harnessing the power of ocean tides -- has become economically feasible as a result of socio-economic problems associated with fossil and nuclear fuels (Stephens and Stapleton, 1981) and improved technology for low-head turbines. The world potential for tidal power production is estimated at 1,240 billion KWh/yr (Gray and Gashus, 1972). At least 50 coastal areas around the world are considered potential sites for tidal power development (Figure 1). This number will increase as the technology of low-head turbines improves, thus increasing the number of coastal areas suitable for tidal power generation. Strong tidal currents associated with shallow coastal environments, similar to those found in a number of southeastern USA estuaries, may be used. Countries that have operational tidal power plants include France, The People's Republic of China, the USSR, and Canada. Only Canada has conducted extensive pre- and post-construction studies to determine potential impact of tidal power on aquatic resources.

Tidal hydropower has advantages over more conventional methods of power generation since it relies on a renewable energy source which is locally available, thus eliminating problems associated with transport of fossil or nuclear fuels to the generating site. No polluting by-products such as noxious hydrocarbons, thermal effluent, or radioactive waste are produced from hydropower plants. Additional benefits include improved infrastructure, flood control, and aquaculture potential (Gray and Gashus, 1972). Tidal hydropower has additional advantages over riverine installations in that it is not influenced by seasonal water levels, floods or droughts (Larsen 1981).

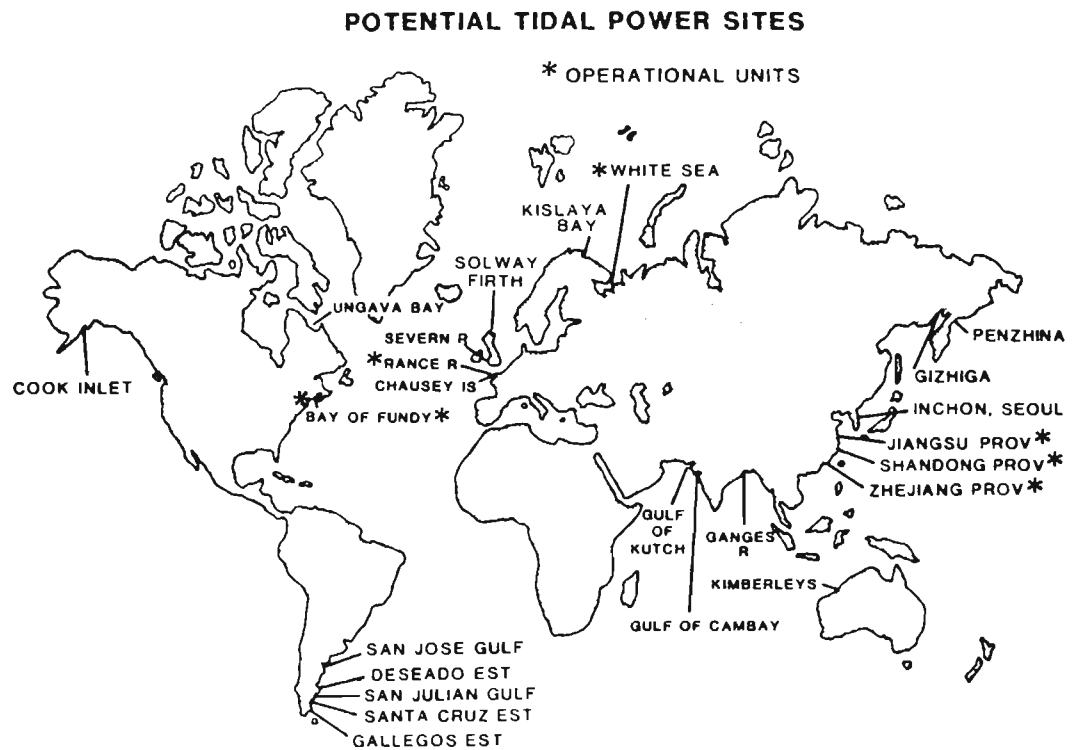


Figure 1. Potential tidal power development sites in the world for the gated barrage design (from Dadswell et al, 1986). Asterisk (*) denotes sites with operational tidal power plants.

The concept of tidal power is not new. Tidal energy was harnessed in Europe as early as the 11th century to produce mechanical energy for exclusive use at the site of generation (Lawton 1972). Tide mills were common up to the mid-19th century in France, Spain, England, Holland, and Russia (Charlier 1982). In North America, Wylie (1979) reported that more than 300 old tide mills were found along the east coast from New Brunswick to Georgia.

As with all electrical power generation, the possible effects on the physical and biological environment must be considered. However, before aquatic ecological effects of tidal power development on southern coastal ecosystems can be predicted, the engineering designs that might be used for ultra-low head power generation must be examined within the context of environmental impact studies conducted on operational tidal power facilities. Results of these studies can be extrapolated to ultra-low head designs that are more feasible for use in the southeastern USA.

GATED BARRAGE DESIGN

Design and Operation

There are several modern designs for tidal power generation. The only design in use is the "gated barrage" type, which is similar in principle to hydropower generation facilities in rivers. This design is used in coastal areas with large tidal amplitude and relies upon the daily or twice daily rise and fall of the tide. A barrage, or dam, containing turbines and sluice gates is constructed across an estuary or embayment to form the headpond. The storage basin is filled at high tide and emptied at low tide

(single effect), or generation takes place during rising and ebbing tides (double effect) against a head which is utilized by turbines of more or less conventional design. The storage basin is typically an estuary or inlet of such shape that a large surface area can be enclosed by a relatively short dam, and the volume of water held by the headpond must ensure production of electricity over a reasonable period. The tidal power facility can use one barrage and basin (single basin) or a combination of barrages and basins (double basin).

There are four basic barrage designs: 1) single basin, single effect; 2) single basin, double effect; 3) double basin, single effect; and 4) double basin, double effect (Figure 2). Detailed descriptions of the modes of operation are provided by Lawton (1972), Wilson and Severn (1972), and Charlier (1982). These designs involve three basic steps (Taylor et al., 1985). The first step, called "sluicing", allows the water to enter the headpond on the flood tide by opening the sluice gates and turbines. At the optimum time, the sluice gates and turbines are closed and the "waiting" step for power generation begins. During this step, the sluice gates and turbine gates are closed until the difference in head between the headpond and the sea is sufficient to begin power generation. The third step is power generation, which is initiated by opening the gates to the turbines.

All modern tidal power plants utilize the single basin configuration. The oldest form of tidal power generation is the single basin, single effect configuration which dates to the tide mills of western Europe in the eleventh century (Charlier, 1982) and was used for production of mechanical energy in North America (Wylie, 1979). The basin is allowed to fill during

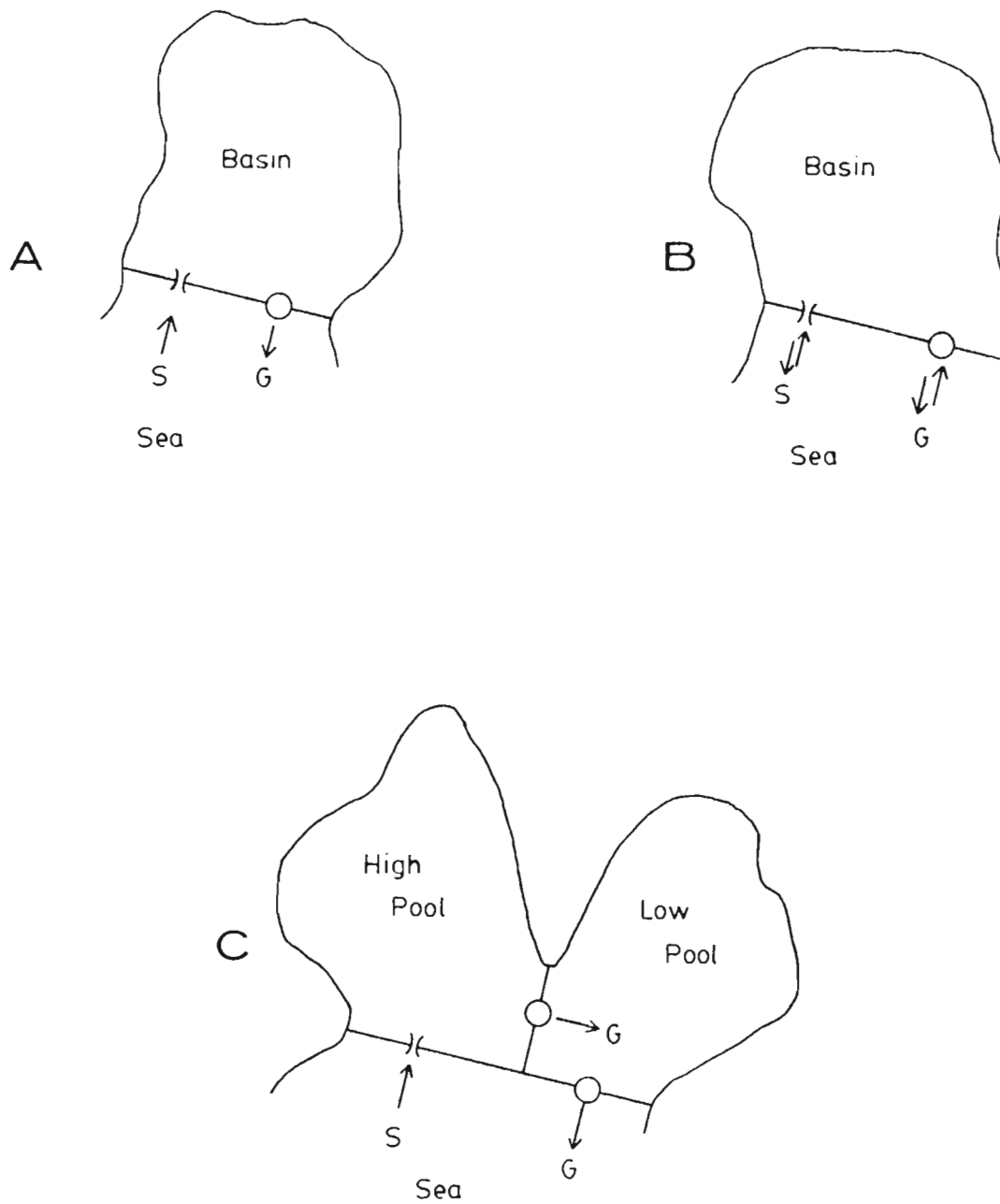


Figure 2. Operational schemes for the gated barrage design:
 A) single basin, single effect; B) single basin,
 double effect; C) interconnected double basin.
 S = sluicing; G = generating.

flood tide through the sluice gates and powerhouse, with turbines spinning freely. Power is generated on the ebbing tide. North America's first modern tidal power plant, the Annapolis Tidal Generating Station in Nova Scotia (Figure 3), is operated in this manner (Douma and Stewart 1981; Dadswell et al., 1984, 1986). The single basin, double effect mode of operation is similar to the Rance River power plant in LaRance, France, which produces power on both flood and ebb tides (Cotillon, 1974). The Rance River plant has an installed capacity of 240,000 kW of power from a head of between 3-13 m with a maximum flow through the plant of approximately 6840 m³/second (Caillez, 1966).

Environmental Impacts

Few studies have been directed toward determining the environmental effects of development and operation of gated barrage tidal power facilities. Results of these studies were summarized by Waller (1972), Shaw (1980), Dadswell et al. (1986) and Rulifson et al. (1986). The majority of these studies addressed problems associated with tidal power development in the Canadian Maritime provinces (Gordon and Dadswell, 1984; Table 1). Fundamental effects for any tidal power design will include changes in primary and secondary productivity of coastal waters, fish mortality, fluctuations in sea level, local weather patterns, and socio-economic structure both locally and regionally (Larsen and Topinka, 1984). Environmental changes may affect natural resources over a wide geographic area (Gordon and Dadswell, 1984) or be restricted to local changes depending on a number of factors such as the size of the basin, the amount of water passed on each tidal cycle, and the natural productivity of the area.

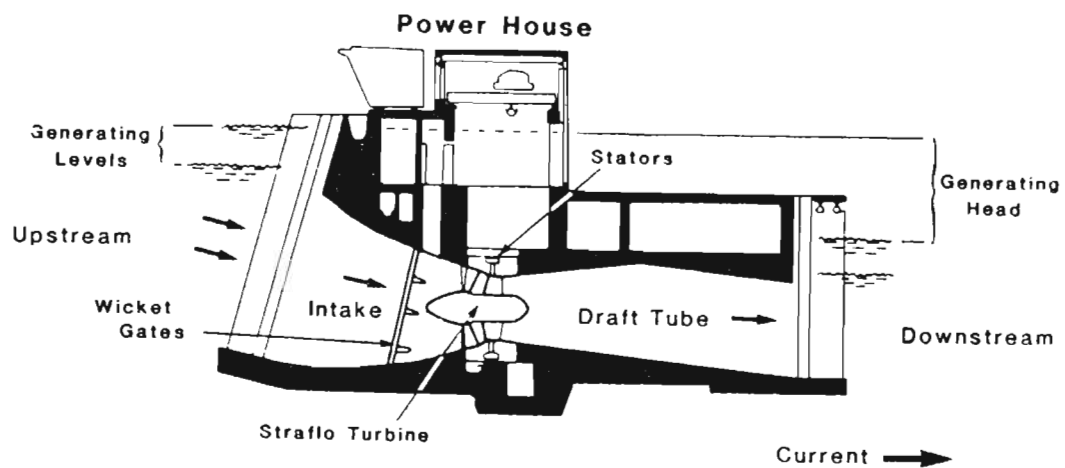


Figure 3. Cutaway view of the Annapolis Tidal Generating Station showing the STRAFLO turbine (from Dadswell et al, 1986)

Table 1. Physical and biological changes associated with tidal power production using barrage type designs under megatidal conditions.

Physical effects

Energy removed from tide
Reduced volume of seawater exchange into basin
Changes in water circulation patterns seaward of and within the basin
Tidal range reduced in basin
Mean water level increased
Increased water column stratification
Decreased turbidity
Increased sedimentation
Changes in saltwater/freshwater boundary -- estuarine and ground water
Reduced storm surge
Increased flushing time

Biological effects

Increased phytoplankton production
Reduced saltmarsh inundation
Shifts in species composition
Shifts in species abundance and distribution
Mortality of biota associated with passage through turbines.

Physical alterations of the environment will occur as a result of removing energy from the tide and reducing the volume of seawater exchange into the basin. These physical changes will ultimately result in alterations of the local ecosystem. Water circulation patterns both seaward of and within the basin will be altered. Within the basin, tidal range will be reduced but mean sea level may increase (Baker, 1984). Reduced energy input to the system will increase vertical stratification of the water column, reduce turbidity, and increase sedimentation in certain areas (Amos, 1984). Increased mean sea level within the basin will effect changes in the saltwater-freshwater boundary within the estuary and within the aquifer (Palmer and Beanlands, 1977). Under normal conditions, an estuary acts as a "shock absorber" for coastal areas during high-energy storms. A barrage-type power generation facility would reduce storm-related damage within the basin. Flushing time of the estuary would be increased, thereby increasing the possibility of effluent disposal and assimilation problems for areas receiving agricultural, municipal, and industrial wastes.

Changes in the biotic system will result from alterations of the physical environment. The most immediate effects are shifts in species composition, abundance and distribution. However, many believe that overall production of benthos and zooplankton will not be altered appreciably by construction of the tidal barrage (Waller, 1972; Daborn et al., 1984; Peer, 1984). Decreased turbidity and increased nutrient load may increase phytoplankton production (Prouse et al., 1984). Increased sedimentation may impact shellfish beds (Risk et al., 1977; Witherspoon, 1984) or change the size and shape of saltmarshes within the system. Changes in the

biological system will be largely controlled by the physical attributes of each site.

The ability to pass fish successfully through tidal power generation facilities is not an easy engineering problem. Millichamp and Staite (1980) suggested gaps in the barrage that would allow up to 5% of the estuarine water to pass unimpeded through the barrage, thus attracting and guiding migrant adult fishes. Ladders and fish lifts are not possible because of the design problems associated with passing fish in two directions (ebb and flood tides). Diversion screens and fish guiding devices such as louvers, lights, or bubble curtains may be feasible for some gated barrage power plants, especially if the number of turbines is small relative to the length of the barrage. Such designs may be applicable to the Annapolis Tidal Generating Station. However, these methods may be impractical for large tidal power dams envisioned for the inner Bay of Fundy and other developments using present engineering concepts. Fish passage must therefore be relegated to open sluice gates and free-spinning turbines in the water control structure on flood tides, and through the turbine during power generation on ebb tides. The critical design factors of conventional turbines for fish passage are: turbine diameter, number of blades, discharge velocity and volume, rotational speed, pressure flux and cavitation potential. These factors were discussed in detail for Kaplan and Francis turbines by Monten (1985), and for the STRAFLO turbine at the Annapolis facility by Dadswell et al. (1986).

"TIDAL FENCE" DESIGN

Design and Operation

A more moderate approach to electrical generation using tidal energy is the concept of employing the tidal wave created by the sun and moon as the energy source rather than a quasi-static hydraulic head created by damming a coastal basin with a large tidal amplitude. This concept involves operation of tidal power plants at ultra-low heads on both flood and ebb tides (double effect) using swift estuarine or coastal currents (Table 2). The barrage structure would be much more open, resembling a bridge rather than a dam, and would possess very large turbine apertures (Figure 4). Power production using the tidal fence design would involve only the power production and sluicing steps of operation; no waiting step is needed. Ocean construction techniques required to build and position barrage modules in deep and fast-flowing water are now available (H.A. Simons Int'l. Ltd., 1984). However, ultra-low head designs are not yet feasible because economically viable turbines have not been developed. The possible options for energy conversion devices in a tidal fence power plant were reviewed by Taylor et al. (1985). Options include both horizontal- and vertical-mount ducted turbines. Because of the very large turbine areas, full gates or sluiceways would be impractical. The tidal fence barrage will always be open to flow, so the entire opening of the estuary must be fitted with flow restrictors (e.g., turbines, partial gates, or other partially or completely impermeable elements) to prevent the water from diverting direction and passing through the areas of least resistance. Navigational locks could be incorporated into the design to facilitate commercial and recreational traffic in navigable waters (H.A.

Table 2. Siting criteria and predicted environmental impact (due to changes in tidal amplitude) of tidal power schemes (after Taylor et al. 1985).

Engineering Design	Usual working head	Basin size	Current velocities	Site Depth	Foundation requirements	Physical Environmental Impact
Gated Barrage						
Single effect or double effect	medium 5-10 m	small	low to moderate	>10 m <30 m	excellent	major local disruption
Open Barrage						
"Tidal fence" (Double effect)	low	large	moderate to high	>10 m <100 m	minimal due to reduced head	moderate disruption
Current turbines ("underwater windmill")	zero (0.05 m)	not applicable	high	>10 m	secure anchorage	negligible

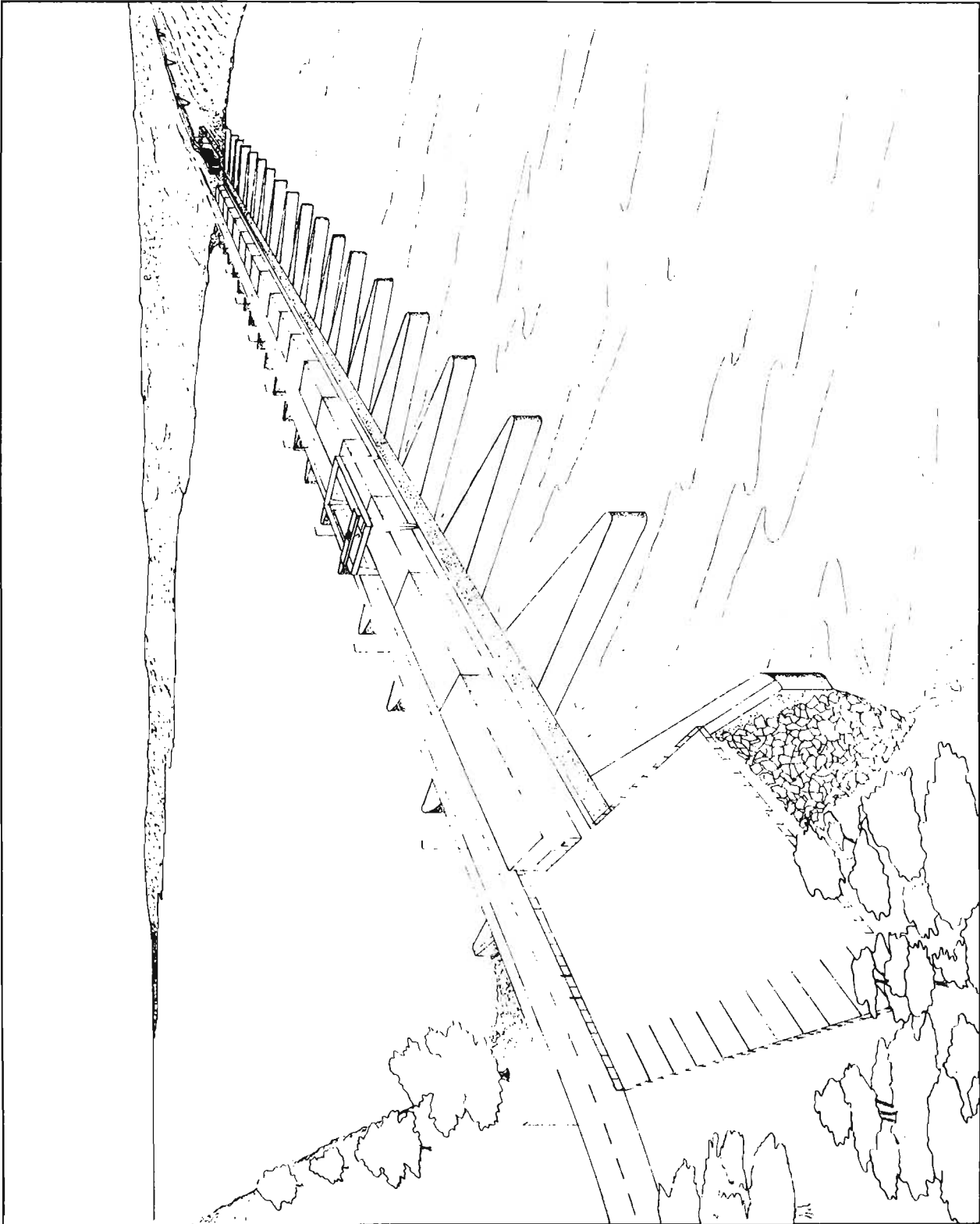


Figure 4. Artist's conception of a tidal fence electrical generation facility in a southeastern USA estuary (courtesy of Tidal Power Corporation).

Simons Int'l. Ltd., 1984). Existing infrastructure for ground transportation could be improved by providing road and rail access over the tidal fence barrage.

Taylor et al. (1985) reviewed the advantages and disadvantages of tidal fence power production relative to the gated barrage design; a number of these points pertain to tidal power development in the southeastern USA. Lower working heads will reduce the hydrostatic load on the environment, resulting in a power plant design with site options for greater variability in channel depth and foundation (bedrock) conditions (Table 2). The open design of the barrage will permit construction in high velocity tidal streams and closure of large basins without severely modifying the natural tidal regime. Energy production can be balanced more equally over the tidal cycle, except at slack water. A major disadvantage is the partial or complete loss of control options (e.g., "waiting") that can be used to augment energy production.

Environmental Effects

Although southern estuaries are much different physically compared to systems normally considered for tidal power development, alterations of their physical and biological environments can be predicted based on results of the gated barrage studies described above. A minimal head is required (about 2 m) for power production, but the head is caused by back-pressure of the water being forced through the tidal fence, not the physical damming and release of water through a gated barrage. This back-pressure will probably reduce tidal circulation within the estuary, thus reducing turbidity and increasing sedimentation in certain areas. Reduced

circulation or altered patterns may decrease the chances of fish and shellfish larvae spawned offshore to be transported successfully to the saltmarsh nursery grounds typical of southeastern estuaries (e.g., Weinstein, 1979; Rulifson, 1983; Miller et al, 1984). Reduced circulation will increase the estuary flushing time, which may increase nutrient loads and cause waste disposal and assimilation problems within the estuary and adjacent waters. Reduced turbidity should increase phytoplankton and zooplankton production in both estuarine and coastal waters. Altered circulation patterns will have unpredictable effects on sedimentation patterns, having a stabilizing effect in some areas while increasing or reducing shoaling problems in other. Large basins may exhibit increased stratification of the water column, which may intensify existing differences in salinities, temperatures, and dissolved oxygen. Storm surge damage within the estuary would be reduced, but the storm energy may be redistributed along the barrier island system creating additional problems of inlet stabilization and shoreline erosion.

Problems associated with fish passage through tidal fence power plants would be similar, though probably less extreme, than those described above for gated barrage facilities. Due to the lack of sluice gates, all fish passage would be accomplished through the turbines and navigational locks on both flood and ebb tides. Since the tidal fence turbines will be much larger, however, the physical stress placed on the fish during passage will be different than for conventional turbines. Estimation of fish mortality or injury during passage through tidal fence turbines has little meaning at this time because full-scale working models of the proposed turbine designs have not been constructed. However, we can predict that the water length

(distance between each successive pass of the runner blades), impact velocity (velocity of a blade striking a fish) and mutilation rates may be similar to that of conventional turbines. Injury caused by pressure flux and cavitation will be reduced because of the low head required for operation.

CURRENT TURBINES

Design and Operation

The current turbine is a concept for generating power from tidal currents using free standing power generation units requiring no barrage and no head (Table 2). The concept is based on wind energy technology and would extract kinetic energy from steady ocean currents using horizontal axis turbines, or from tidal currents using vertical axis turbines. Power generation would occur through most of the tidal cycle, initiated when a minimal velocity is present at the start of ebb or flood tide and terminated when the water velocity dips below the minimum (Taylor et al. 1985). Such a power plant facility would resemble a series of underwater windmills positioned in the tidal channel of an estuary. This type of design would allow only a fraction of the kinetic energy available to be intercepted and captured and would not be as efficient as the barrage type designs described above.

The advantages of such a design in protecting the environmental integrity of southeastern USA estuaries is obvious. Environmental impact to the physical and biological systems would be reduced. Mortality of fish and shellfish larvae immigrating to estuarine nursery grounds, and of migrant adults to spawning areas, would be lower than for conventional

turbines. No cavitation or pressure mortality is expected; death caused by blade strikes may reduce mortality by one-half to two-thirds. Obstruction to navigation would be minimal, except for the substantial foundations and structural supports required for holding the power unit in place. Effects on water circulation patterns, sedimentation, etc. would be negligible relative to conventional tidal hydropower but may cause problems for organisms utilizing tidal transport to enter (Miller et al. 1984) or exit (Rulifson 1983) estuaries.

SUMMARY

Electrical generation facilities utilizing tidal energy are being developed around the world. As technology for electrical generation using ultra-low head turbines improves, the number of sites available for tidal power development will increase to include estuaries and coastal areas of the southern USA possessing strong tidal currents. Three designs of tidal power facilities are described, one of which is in use. The other two are feasible for use in southeastern USA but waters the technology is not fully developed. The gated barrage design is used in coastal areas with large tidal amplitude. A moderate head of 8-10 m is created by damming estuarine waters and generating power with conventional turbines on ebbing tides. The estuary is filled on flood tides by passing water through sluice gates and the powerhouse. Major alterations to the physical and biological systems occur with this design. The tidal fence concept would utilize large turbines positioned in a structure resembling a bridge or causeway. No sluice gates are required with this design. All migrant biota would pass through the turbines on both ebb and flood tides, but injury caused by

cavitation and pressure flux would be minimal. The current turbine design would be the most environmentally compatible but least efficient in power production. Current turbines would resemble underwater windmills and utilize wind energy technology for power generation. No dam or head is required, obstruction to navigation is negligible, and biota would pass relatively unimpeded to and from estuarine habitats to the adjacent coastal waters.

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MITIGATION BANKING IN THE SOUTHEAST¹

James D. Brown
U.S. Fish and Wildlife Service
Atlanta, Georgia

David M. Soileau
U.S. Fish and Wildlife Service
Lafayette, Louisiana

R. Wilson Laney
U.S. Fish and Wildlife Service
Raleigh, North Carolina

ABSTRACT

In recent years, a concept known as "mitigation banking" has evolved as a mechanism for achieving mitigation of unavoidable habitat losses associated with water resources development projects. In mitigation banking, habitat improvement actions are conducted in advance of project construction. Mitigation credits attributable to these actions are placed in a mitigation bank account and can be used later to compensate unavoidable habitat losses resulting from actual project construction. This innovative approach is being used with some success, mainly on the West Coast and in the Southeast. Two good examples of mitigation banks in the Southeast are a 5,000-acre bank established by the Tenneco Oil Company for mitigation of the impacts of oil exploration and development in coastal Louisiana wetlands, and a 1,436-acre bank established in the lower Roanoke River Basin by the North Carolina Department of Transportation for mitigation of the impacts of highway projects on bottomland hardwoods. Both banks are associated with wetlands permitting activities and are being used to accomplish desirable habitat preservation and management goals.

¹The findings, conclusions, opinions, or recommendations expressed in this paper are those of the authors and do not necessarily reflect the views or positions of the U.S. Fish and Wildlife Service.

INTRODUCTION

Mitigation of fish and wildlife habitat losses caused by water resources development activities is an issue that increasingly has occupied the attention of managers, planners, and developers. The term generally is considered to mean avoiding or minimizing adverse impacts to fish and wildlife and compensating for unavoidable losses of those resources. Historically, mitigation has been one of the most troublesome aspects of planning for water resources development projects because it came at the end of the permitting process. Disagreements over the definition of mitigation, how best to mitigate, how much mitigation is required, and even who is responsible for mitigation have been commonplace. As a result, many projects have been planned and constructed with little or no mitigation, and valuable fish and wildlife habitats have been degraded or lost without compensation. Loss of habitat is the most critical fish and wildlife problem in the United States, and the mitigation of these losses whenever possible is absolutely imperative.

In an effort to improve the mitigation process, the U.S. Fish and Wildlife Service (1981) promulgated a formal mitigation policy to guide its involvement in the planning of water resources development projects. This policy presented a definition of mitigation, established Resource Categories of fish and wildlife habitat with associated mitigation goals, and described procedures for quantitatively determining mitigation needs based on the value of the affected habitat for fish and wildlife. One approach to achieving mitigation identified as acceptable in the policy is "mitigation

banking." This innovative approach to mitigation has been employed with some success in recent years (Soileau et al. 1985; Niedzialkowski and Jaksch 1986), mainly on the West Coast and in the Southeast. Approximately 12 mitigation banks are in various stages of planning and operation in the United States.

Four mitigation banks exist in the Southeast. These include a bank established by the Tenneco Oil Company in south central Louisiana for mitigation of the impacts of oil exploration and development in coastal Louisiana wetlands, and a bank established in the lower Roanoke River Basin by the North Carolina Department of Transportation for mitigation of the impacts of highway projects on bottomland hardwood wetlands. The two other banks are a small tidal marshland bank in Chesapeake, Virginia, operated by the Virginia Department of Highways and Transportation (Russell 1983), and a bottomland hardwoods bank in Louisiana operated by the Louisiana Department of Transportation and Development. Brief descriptions of the Tenneco and North Carolina Department of Transportation banks are included in this paper for the purpose of illustrating the concept of mitigation banking.

MITIGATION BANKING CONCEPT

Mitigation banking is defined in the Service's mitigation policy as "habitat protection or improvement actions taken expressly for the purpose of compensating for unavoidable, necessary losses from specific future actions." Soileau et al. (1985) provided the following generalized explanation of mitigation banking:

"In simplified terms, mitigation banking is similar to maintaining a bank account. A developer undertakes measures to create, restore, or preserve fish and wildlife habitat in advance of an anticipated need for mitigation for project construction impacts. The benefits attributable to these measures are quantified, and the developer receives mitigation credits from the appropriate regulatory and/or planning agencies. These credits are placed in a mitigation bank account from which withdrawals can be made. When the developer proposes a project involving unavoidable losses of fish and wildlife resources, the losses (debits) are quantified using the same method that was used to determine credits, and a withdrawal equal to that amount is deducted (debited) from the bank. This can be repeated as long as mitigation credits remain available in the bank."

The Fish and Wildlife Service's mitigation policy incorporated the definition of mitigation contained in the Council on Environmental Quality's National Environmental Policy Act regulations (40 CFR Part 1508.20[a-e]). By that definition, mitigation can include:

- (a) avoiding the impact altogether by not taking a certain action or parts of an action;
- (b) minimizing impacts by limiting the degree or magnitude of the action and its implementation;
- (c) rectifying the impact by repairing, rehabilitating, or restoring the affected environment;
- (d) reducing or eliminating the impact over time by preservation and maintenance operations during the life of the action; and,
- (e) compensating for the impact by replacing or providing substitute resources, or environments.

This definition recognizes both the modification of project design to avoid or lessen impacts and compensation for impacts that are unavoidable. Avoiding the impact altogether is the preferred mitigation method, followed by minimizing the impact. These may be accomplished by such measures as relocating the project to a less environmentally sensitive site, selecting a project plan that avoids or lessens adverse impacts, or modifying the project plan. Compensation, which involves actions taken to offset unavoidable impacts, is the least favored mitigation method and should be considered only after all feasible means of avoiding or reducing impacts have been exhausted. Compensation can involve a variety of habitat creation, restoration, or management activities. By its definition, mitigation banking is a mitigation approach that can be used only when compensation is warranted.

An important element in mitigation banking is the need for a standardized, replicable quantitative method of determining credits and debits. The Habitat Evaluation Procedures (HEP) developed by the U.S. Fish and Wildlife Service (1980) provide a good method of doing this. HEP is founded on the premise that habitat can be evaluated in terms of its ability to support specific fish and wildlife populations (evaluation elements). HEP provides a means of evaluating existing conditions and future conditions with and without a project. Thus, the benefits resulting from habitat improvement actions can be quantified through a HEP analysis and these benefits then constitute the bank account available to a developer. As individual development projects occur, their impacts can be similarly quantified and deducted from the bank account. HEP is not the only acceptable method of quantifying credits and debits; other credible evaluation systems may be used. In some cases, credits and debits may have to be determined using the best professional judgment available.

TENNECO OIL COMPANY MITIGATION BANK

In October 1982 the Tenneco Oil Company proposed the establishment of a mitigation bank on approximately 7,000 acres of fresh to brackish marsh (palustrine emergent wetlands to estuarine intertidal emergent wetlands, Cowardin et al. 1979) in Terrebonne Parish in south-central Louisiana. Federal and state agencies that participated in the planning of that bank included the Fish and Wildlife Service, the National Marine Fisheries Service, the Soil Conservation Service, the Louisiana Department of Wildlife and Fisheries, and the Louisiana Department of Natural Resources.

The mitigation bank site lies within an area that is undergoing a rapid loss of coastal wetlands. The rate of coastal marshland loss in Louisiana exceeds 40 square miles per year and is increasing (Wicker 1980; Gagliano 1981). These losses have resulted from a combination of natural and man-induced causes, including rising sea level, subsidence, the physical processes of growth and deterioration of the Mississippi River Delta, channelization and levying of the Mississippi River, excavation of canals for oil and gas exploration, and extraction of groundwater. The flow of nutrient and sediment- rich fresh waters to these marshes has been blocked or diverted. Combined with subsequent intrusions of salt water, this has resulted in conversions of large areas of fresh marsh to brackish or saline marsh or to estuarine open water. Records indicate that the original fresh marshes of the mitigation bank site have undergone such

conversion to brackish marsh and open water due, in part, to the elimination of fresh water and sediment inflow in the early 1900's. This deterioration was projected to continue unless appropriate management steps were undertaken.

The marshes and shallow open waters of the mitigation bank site provide excellent habitat for numerous adult and juvenile finfishes and shellfishes. Because of the diverse salinity regimes, both freshwater species (largemouth bass, bluegill, blue catfish, and channel catfish) and estuarine species (Gulf menhaden, seatrout, croaker, blue crab, white shrimp, and brown shrimp) utilize the area. The wildlife value of the area is considered equally high. The area consistently winters a tremendous number of waterfowl, and there is keen competition for hunting leases. Non-game wetland birds that occur in the bank site include egrets, herons, ibises, and white pelicans. The area also supports an abundant population of furbearers. Large numbers of nutria, muskrat, and alligators are harvested on the site, and smaller numbers of raccoon and mink are taken.

Tenneco proposed to implement a structural water management plan that would reintroduce fresh water and sediment inflow, improve water circulation, and reduce intrusion of salt water. This management program was projected to reduce significantly the rate of marsh loss and to extend the projected life of the marsh even after the allowable credits and been debited. An interagency team measured the

anticipated benefits of the proposal using HEP. The lengthy and intensive negotiations that ensued culminated in an interagency Memorandum of Agreement (MOA) in January 1984, which formally established the mitigation bank. The MOA included a description of the bank's purpose, general recommendations, and 18 provisions governing the implementation and operation of the bank. Some of the more important provisions in the MOA are listed below.

1. The interagency review team, here and after referenced, consisting of the U.S. Fish and Wildlife Service (serving as chairman), the U.S. National Marine Fisheries Service, the U.S. Soil Conservation Service, the Louisiana Department of Natural Resources, and the Louisiana Department of Wildlife and Fisheries, shall determine habitat units and the AAHU's¹ to be initially credited to the mitigation bank and shall determine future debits and credits to the mitigation bank.
2. The appropriate parties to this Agreement shall use HEP, or a mutually agreeable and credible methodology, to determine credits and debits to be applied to the mitigation bank.

¹ Average Annual Habitat Units, a measure of habitat value calculated in HEP.

3. Mitigation by debiting available AAHU's from the mitigation bank is appropriate and will be used to offset only unavoidable impacts on fish or wildlife when the applicant can demonstrate to the satisfaction of all parties to this Agreement that there are no onsite alternatives which are available and capable of being done after taking into consideration cost, existing technology, and logistics in light of overall project purposes.
4. Credits generated within Hydrologic Unit 5 shall be applied to activities requiring mitigation within that same Hydrologic Unit and may be applied outside of Hydrologic Unit 5 only with the approval of the interagency review team. In no case shall credits be applied to projects outside the State of Louisiana.
5. Tenneco may buy, sell, trade, or otherwise dispose of mitigation credits in the form of AAHU's to be debited from the mitigation bank. The buyer or assignee of such AAHU's may use such credits to satisfy its mitigation obligations subject to applicable laws, regulations, and provisions of this Agreement. The interagency review team shall be informed prior to such AAHU transfers.

6. As the Tenneco proposal is a pilot program, 5 years after implementation of the management plan a complete evaluation of the management program shall be made by the interagency review team using HEP, or a mutually agreeable and credible methodology. A preliminary assessment is to be made 1 year after implementation of the management program to evaluate the effectiveness of the management program. Whenever significant operational and/or structural changes are made to improve success, another complete evaluation shall be made in 3 to 5 years following those changes.

The mitigation bank was established on 5,000 acres of land owned by Tenneco, with Tenneco retaining ownership of the property. Another adjacent 2,000 acres of coastal wetlands under other private ownership that are receiving benefits from the management plan may be incorporated into the bank in the future. The details of a complete analysis of the Tenneco bank, including a copy of the MOA, are contained in a report by Soileau (1984).

After the bank was established in early 1984, Tenneco obtained the required state and Corps of Engineers permits and installed the necessary water management structures. Since that time, use of the bank as mitigation credit for 10 permit applications for canal excavation has been proposed. The debiting process has been delayed by ongoing discussions about the number of credits that should be withdrawn from the bank account. When agreement is reached on this question, the mitigation bank account will be debited for the first time.

NORTH CAROLINA DEPARTMENT OF TRANSPORTATION MITIGATION BANK

The North Carolina Department of Transportation entered into discussions with the Fish and Wildlife Service, the North Carolina Wildlife Resources Commission, and the North Carolina Nature Conservancy in the spring of 1985 concerning the establishment of a mitigation bank in the lower Roanoke River Basin. This bank was proposed for the purpose of mitigating unavoidable impacts to bottomland hardwoods habitat (palustrine forested wetlands, Cowardin et al. 1979) expected from future highway and bridge construction in North Carolina. These discussions went very smoothly because the interagency group was able to draw upon the experience gained from the Tenneco bank process and to adapt much of the language in the Tenneco MOA to its needs. These discussions culminated in an interagency Memorandum of Understanding (MOU) signed by the four participants in September 1985.

The MOU established the mitigation bank on a 1,436-acre tract of land known as Company Swamp adjacent to the lower Roanoke River in Bertie County. The North Carolina Legislature appropriated funds specifically for the purchase of lands to be used as a mitigation bank, and the Company Swamp tract was acquired through the cooperative efforts of the State of North Carolina and the North Carolina Nature Conservancy. Through this process, the North Carolina Wildlife Resources Commission gained 100 percent ownership of approximately 700 acres of selectively cut-over bottomland forest and approximately 44 percent interest in the remaining 736 acres which presently exist as a

climax gum-cypress forest. The North Carolina Wildlife Resources Commission has management responsibility for the entire tract in perpetuity.

The entire Company Swamp tract contains some of the finest alluvial bottomland forests in North Carolina and the Southeast. These forests provide food, water, breeding territories, and cover for waterfowl, furbearers, wading birds, songbirds, and numerous small mammals, amphibians, and reptiles. They also provide valuable escape and foraging habitat for larger species, such as whitetail deer, black bear, and wild turkey. The area is frequently flooded and is used by anadromous blueback herring as forage and spawning sites.

The MOU for the North Carolina Department of Transportation bank was patterned after the Tenneco MOA, and several of its 12 operational provisions are identical or similar. Provisions were included to designate the geographic area in which the bank can be applied, to provide for the determination of credits and debits by an interagency team, to restrict use of the bank for only unavoidable impacts, and to provide for monitoring and reevaluation of the project. The agreement also provides for the use of HEP to determine debit requirements, but only for projects larger than 5 acres. Mitigation for smaller projects will be on an acre-for-acre basis. Another notable difference between this bank and the Tenneco bank is that the North Carolina agreement does not allow the buying and selling of mitigation credits. This bank was planned specifically to mitigate losses of bottomland hardwood wetlands in North Carolina; consequently the MOU contains a detailed definition of bottomland hardwood wetlands.

An interagency team has been working on a HEP analysis over the past several months to determine baseline values for the mitigation bank area. Field sampling was completed in August 1986. A report indicating the number of mitigation credits available in the bank account will be submitted to the North Carolina Department of Transportation in February 1987. The North Carolina Wildlife Resources Commission has not yet developed a final management plan, but tentative management measures are being identified and evaluated in order to determine the number of preliminary credits by the February 1987 due date. Seven projects of less than 5 acres already have been applied against the bank account. Two larger projects that require a HEP analysis are awaiting action.

MANAGEMENT CONSIDERATIONS

Mitigation banking provides a mechanism by which unavoidable fish and wildlife habitat losses associated with certain water resources projects can be compensated. This is only one approach to mitigation and is applicable only in limited circumstances. Most mitigation banks have been used for permitted development activities affecting wetlands, although a bank could be established for other habitat types. In general, a mitigation bank should be considered only when habitat improvement activities, such as wetlands creation, restoration, or management, create quantifiable benefits to fish and wildlife habitat value that can be placed in a mitigation bank account. A simple transfer of ownership of land to a public agency does not constitute mitigation and, therefore, would not qualify for establishment of a bank.

The desirability of mitigation banking derives more from the standpoint of administration and practicality than from its biological advantage. Mitigation banking inherently involves offsite mitigation. From a biological standpoint, onsite mitigation is preferable. Onsite mitigation maintains habitat and biological diversity within the landscape and comes closer to maintaining historical patterns of wildlife abundance and distribution. Unfortunately, providing follow-up and management of the small, widely scattered tracts of habitat created through onsite mitigation is difficult. Mitigation banking offers a solution to the problem by providing one large, more easily managed tract.

The most likely candidates for use of a mitigation bank are development interests that have predictable, recurring development needs, such as port authorities, highway agencies, and petroleum development companies. Mitigation banking has not been used for power generation projects, although there appears to be a potential for application where projects result in direct habitat alteration or destruction. Because power plants require siting near a cooling water supply, wetlands often are affected. Since large land areas frequently are acquired for a plant site, it might be feasible to dedicate and operate a wetlands area as a mitigation bank. The application of mitigation banking to other impacts of power projects, such as entrainment and impingement of aquatic organisms and thermal effects, has not been explored and would require thoughtful analysis to develop an acceptable approach.

A mitigation bank should be planned and organized in a simple, straight-forward manner. Ideally the bank should be established and administered by a formal interagency agreement, such as those used for the Tenneco and North Carolina Department of Transportation banks. The agreement should clearly specify the geographic area in which the bank can be applied, the party responsible for serving as the banker, the party responsible for conducting habitat improvement and management actions, the methodology for determining credits and debits, the types of impacts for which the bank can be used, the monitoring and reevaluation responsibilities, and the life of the

bank. Specific guidance on these considerations has been presented by several authors (Soileau et al. 1985; Zagata 1985; Dunham 1986; Niedzialkowski and Jaksch 1986; Riddle and Denninger 1986); however, specific provisions may vary considerably from one mitigation bank agreement to another.

Mitigation banking can provide benefits to both developers and fish and wildlife resources. For the former it saves time and money by incorporating mitigation into the early planning process, although the time required for a HEP analysis sometimes can cause delays. It also results in improved public recognition and credit being given to developers for working cooperatively with fish and wildlife agencies to undertake needed management actions. For wildlife, mitigation banking assures that mitigation for unavoidable losses will occur. It also fosters mitigation approaches that can be incorporated into basin or estuary-wide management programs. In the past, many small wetlands mitigation projects have not been successful (Fehring 1983; Dial et al. 1985; Race 1985). In contrast, mitigation banking can offer an opportunity to plan larger mitigation projects designed to accomplish long range fish and wildlife goals, such as restoring specific vegetation types, increasing habitat diversity, or providing habitat for target species. Mitigation banking also can include follow-up and monitoring requirements to ensure that mitigation measures are successfully implemented.

Substantial risks are associated with mitigation banking. A major risk is the potential neglect of good project planning. For example, wetland regulatory agencies that are charged with protecting the

public's interest in wetlands are also under a great deal of pressure to approve permits. Quick reliance on the mitigation bank may provide an easy way out of the dilemma. To avoid this, use of the mitigation bank must be allowed only for truly unavoidable habitat losses, after all reasonable efforts to avoid or minimize losses have been made. Moreover, other planning and regulatory procedures and requirements, such as water dependency, availability of alternative sites, and a public interest determination, must not be omitted or neglected. A mitigation bank also must be operated in a manner that will instill public confidence, since it can lead to a perception that permits are being bought and sold. Most of these problems can be avoided by careful interagency cooperation and planning in the development of a mitigation bank.

CONCLUSION

The alteration or destruction of habitat, particularly wetlands, caused by water resources development projects is one of the most serious problems faced by fish and wildlife managers. For the most part, efforts to obtain adequate mitigation of these impacts have been largely unsatisfactory. In recent years, mitigation banking increasingly has been used as a mechanism for achieving mitigation for unavoidable habitat losses. This approach has attracted widespread attention and has stimulated considerable discussion. Mitigation banking offers both benefits and disadvantages and can be used only for certain types of projects and impacts. However, if administered properly, the approach can provide an effective means of meeting mitigation needs.

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SUCCESSFUL LICENSING OF A MAJOR POWER COMPLEX
IN THE SOUTHEASTERN UNITED STATES
ENCOMPASSING MULTIPLE REGULATORY AGENCIES GOALS

Charles J. Zimmerman, Jr.
Lyle A. Lydick
Dames & Moore
455 East Paces Ferry Road
Suite 200
Atlanta, Georgia 30363

ABSTRACT

This report provides an overview of the planning effort and the implementation of the various steps during a several year period for constructing a large power complex in the Southeastern United States. South Carolina Electric & Gas Company (SCE&G) planned and carried out all of the activities associated with licensing and constructing both a major pumped storage hydroelectric facility and a nuclear power plant. In addition, the utility updated an existing power plant to accommodate increased water needs by the new facilities. The objectives of the owner and of the regulatory agencies were fulfilled.

Extensive planning and coordination between the involved agencies were necessary in order to satisfy the various regulatory requirements to license these facilities. Requirements were met for every phase of the program. Studies were conducted for environmental baseline descriptions of the project areas; construction and operation monitoring also took place as required. The combined uses by these two new facilities of a common reservoir required innovative study approaches to isolate impacts associated with each component. The biological studies had to be carefully planned in order to satisfy each agency's unique study requirements.

In response to the regulatory requirements of several state and federal agencies involved in the decision-making process for the license applications for these two plants, a sampling scenario was developed that estimated impacts associated with each facility. Complications of operating these two programs simultaneously will be described.

Concurrent with performance of the required environmental studies, mitigative measures were employed at various stages of the project. For example, since the pumped storage facility came on line approximately six years before the nuclear station, mitigative measures such as management programs to enhance the area for waterfowl and fish were in place prior to completion of the construction of the nuclear facility. These wildlife management programs were Federal Energy Regulatory Commission (FERC) license requirements.

In general the schedules were met for both of the facilities. The pumped

storage facility was completed within schedule and the nuclear power plant was delayed only slightly in its overall construction and operation.

Although the generating facilities were developed primarily for use of SCE&G, energy was available to assist other systems in the case of an emergency. South Carolina Electric & Gas Company is a member of the Southeastern Electric Reliability Council. Other members are the major utilities and cooperatives in the southeastern United States.

Today, the facilities are operating and providing electrical energy resources to the public. The mitigative measures that were employed are continuing to be effectively managed as required. These water resources have been widely utilized by the residents of nearby communities and other visitors to the project vicinity. In conclusion, the overall process of environmental planning, documentation, and implementation of biological studies as well as mitigation programs were well thought out and executed according to schedule. This resulted in a cost efficient environmental program supporting the licensing and operation of the facilities while meeting the needs of regulatory agencies and, more importantly, the public.

INTRODUCTION

It takes approximately 12 to 14 years to plan, license, and construct a power generation station. During that time period, a siting study is completed, consultation with several regulatory agencies is completed, and a license is granted by the lead regulatory agency. Prior to granting a permit or license, scientific investigations are carried out for the various disciplines, including geotechnical, cost evaluation, and environmental. This paper describes the environmental studies.

The objectives of the owner were to:

- Construct and license two generating stations to meet future generation needs.
- Complete the project within the specified time and within budget.

The objectives of the agencies are:

- Mitigate habitat loss. As protectors of the environment, the agencies' charge was to ensure that the resources would remain for future generations.
- Ensure that project discharge waters were within acceptable limits.
- Offset fishery losses in the existing reservoir.

Site Description

The project site in Fairfield and Newberry Counties is located in the Broad River watershed about 26 miles northwest of Columbia, South Carolina. The site includes a hydroelectric plant, a pumped storage hydroelectric facility, and a nuclear generating station. The Broad River (6,100 cfs average annual flow) is one of the major headwater tributaries of the Santee River Basin and has its origin in the Blue Ridge Mountains of North Carolina.

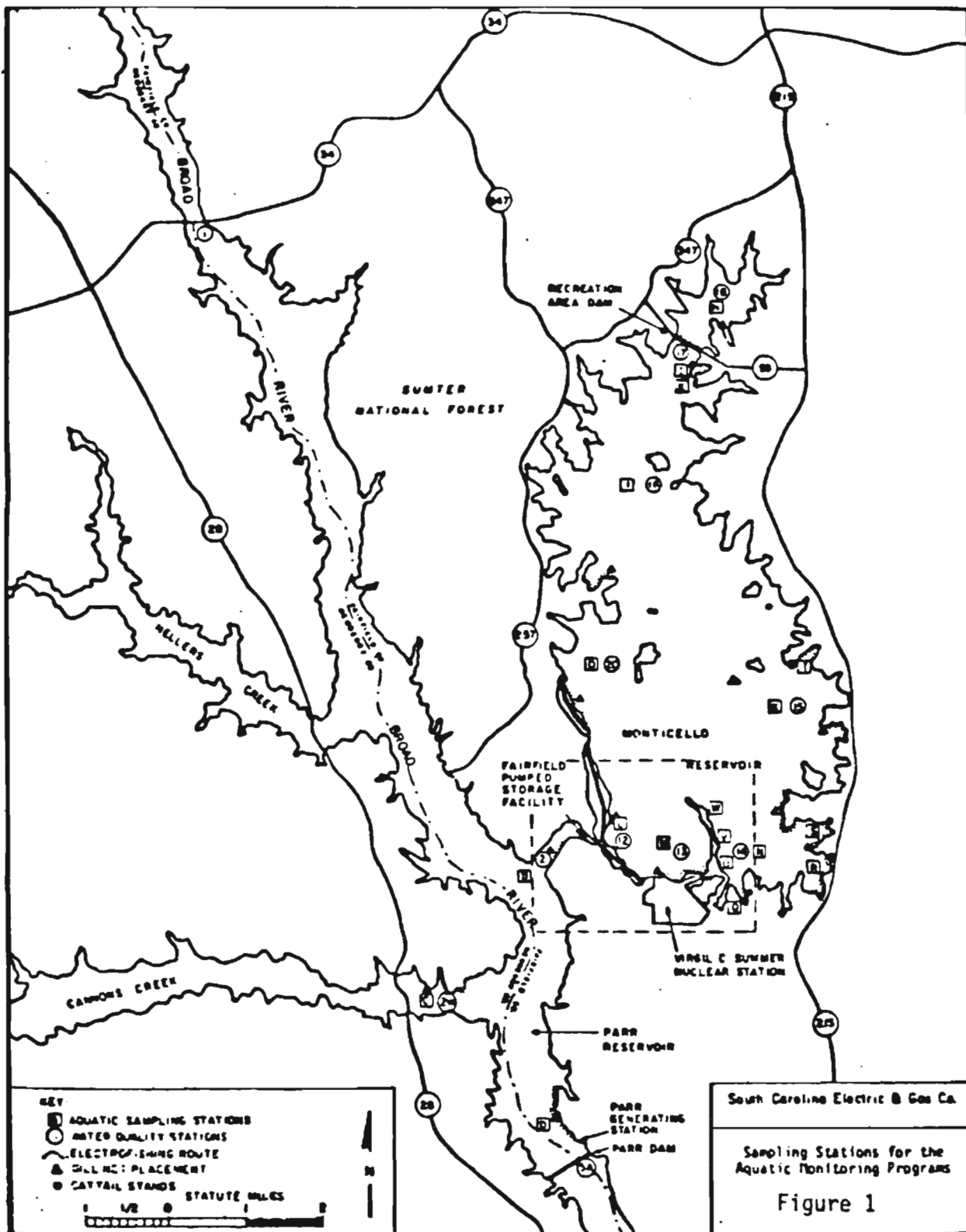
The planned approach to development of the generating facilities was to be completed in phases. The first phase included updating the facilities at the existing Parr Hydroelectric Generating Station.

The Parr Hydroelectric Generating Station with a capacity of 15 megawatts became operational in 1916. It has generated electricity for SCE&G customers continuously since that time, with the exception of

being shut down occasionally for maintenance and repairs. To complete the planned power complex the existing dam at the Parr Generating station had to be raised. The increased dam height was necessary to increase storage capacity in the lower pool for the proposed Fairfield Pumped Storage Facility. An engineering evaluation determined that raising the existing dam a total of 12 feet by the installation of a series of Bascule gates, would be the most cost-effective approach to this problem. This would allow a usable storage capacity of 29,000 acre-feet in the lower pool (Figure 1).

The second phase of the project was to complete the 480 MW Fairfield Pumped Storage Facility (FPSF). This generating station provides power for peak loads at certain times of the day; with the increased usable storage in the lower reservoir this station can generate for 8 hours. There are eight reversible pump-turbine units, each capable of producing 60 MW of electricity. These turbines receive water from the upper reservoir via four penstocks, each 26 feet in diameter. There is approximately 150 feet of head between the upper and lower pools. The units came on line in 1978.

The final phase of the project was the completion of the Virgil C. Summer Nuclear Station. This station was constructed for generating electricity for the base load of the SCE&G's territorial system. The nuclear steam supply system for the Virgil C. Summer nuclear station is supplied by a pressurized Westinghouse Electric Corporation reactor;



the turbine generator has a power output of 900 MW. Cooling water is withdrawn from the 6,800-acre Monticello Reservoir at the rate of 534,000 gpm, passed through the system, and returned to Monticello Reservoir. This reservoir was built to provide cooling water for the nuclear station and serves as the upper pool for the pumped storage facility.

REGULATORY REQUIREMENTS

In response to the regulatory requirements of state and federal agencies involved in the decision making process for approval of the license application for the two proposed facilities, a sampling scenario was developed that complied with agency requirements, and that evaluated impacts associated with each facility.

Environmental Monitoring

The environmental monitoring program for the projects began with data collection requirements stipulated by the FERC. Baseline monitoring of water quality, aquatic, and terrestrial resources began in 1971 on the Broad River and Parr Reservoir (Figure 2).

Aquatic baseline monitoring continued through the Fall 1973 at stations on the Broad River and Parr Reservoir. Terrestrial biology monitoring at selected locations was also carried out. During 1974 through 1977 construction monitoring for aquatic and terrestrial resources was carried out.

<u>Program</u>	<u>Component</u>	<u>1971</u>	<u>1972</u>	<u>1973</u>	<u>1974</u>	<u>1975</u>	<u>1976</u>	<u>1977</u>	<u>Location</u>
Aquatic	Vascular Hydrophytes	A	A	A					Broad River and Parr Reservoir
	Plankton	Q	Q	Q	Sp Su F	Sp Su F	Sp Su F	Sp Su F	
	Benthic	Q	Q	Q	Sp	Sp	Sp	Sp	
	Macroinvertebrates				Su F	Su F	Su F	Su F	
	Fish	Q	Q	Q	Sp Su F	Sp Su F	Sp Su F	Sp Su F	
Terrestrial	Soil		A						Project Area
	Vegetation	Q	Q		A	A	A	A	
	Invertebrates		A		A	A	A	A	
	Amphibians & Reptiles	Q	Q	S					
	Birds	Q	Q	Q	W Su	W Su	W Su	W Su	
	Mammals	Q	Q				F	F	
<hr/>									
A = Annual		Sp = Spring							
Q = Quarterly		Su = Summer							
W = Winter		F = Fall							

FIGURE 2. SUMMARY OF BIOLOGICAL MONITORING PROGRAMS, 1971-1977.

Monticello Reservoir reached full pool level in February 1978 and operational monitoring for the Fairfield Pumped Storage Facility began in June 1978 (Figure 3). The monitoring programs were designed to meet the licensing requirements of the Federal Energy Regulatory Commission, and requirements of the National Pollutant Discharge Elimination System Permit issued by the South Carolina Department of Health and Environmental Control (SCDHEC). These studies provided a monthly assessment of changes in the ecosystem, the results of the studies allowed the comparison of baseline conditions and operations impacts.

Beginning in January 1983 post-operational monitoring programs were designed to meet the licensing requirements of the Nuclear Regulatory Commission and the requirements of the National Pollutant Discharge Elimination System permit issued by the South Carolina Department of Health and Environmental Control (Figure 4). The purpose of the monitoring program was to assess the thermal effects associated with the heated effluent from the V.C. Summer Nuclear Station on the biota in Monticello Reservoir.

Mitigation

Due to the unavoidable adverse impact of constructing and operating a power facility such as the V.C. Summer Nuclear Station/Fairfield Pumped Storage Facility generating complex, measures were taken to minimize these effects. Minimization of the construction impacts included erosion and dust reduction measures and borrowing fill

VIRGIL C. SUMMER NUCLEAR STATION
AND FAIRFIELD PUMPED STORAGE PROJECTS

		1978	MAY	JUNE	JULY	AUG.	SEPT.	OCT.	NOV.	1978 DEC.
LOCATION	BIOLOGICAL STATION	BEGIN PREOPERATIONAL STUDIES MODES (DNCE), FERC								
	WATER QUALITY STATION	WATERFLOW SURVEY (F) SONG BIRD AND UPLAND GAME BIRD SURVEY (F)								
	SAMPLE TYPE	SURVEY, MAP AND IDENTIFY VASCULAR HYPOPHYTES (F) PARR AND MONTICELLO RESERVOIRS								
	REQUIRED BY	D - DNCE F - FERC								
		INFRARED PHOTOS LATE 2ND QUARTERLY SAMPLE 3RD QUARTERLY SAMPLE 4TH QUARTERLY SAMPLE								
PARR RESERVOIR STATIONS	B	2	PHYTOPLANKTON							
			ZOOPLANKTON							
			MACROINVERTEBRATES & FISH							
			WATER QUALITY							
	C	C	PHYTOPLANKTON							
			ZOOPLANKTON							
			MACROINVERTEBRATES & FISH							
			WATER QUALITY							
	D	D	PHYTOPLANKTON							
			ZOOPLANKTON							
			MACROINVERTEBRATES & FISH							
			WATER QUALITY							
MONTICELLO RESERVOIR STATIONS	I	17	PHYTOPLANKTON							
			ZOOPLANKTON							
			MACROINVERTEBRATES & FISH							
			WATER QUALITY							
	J	16	PHYTOPLANKTON							
			ZOOPLANKTON							
			MACROINVERTEBRATES & FISH							
			WATER QUALITY							
	K	15	PHYTOPLANKTON							
			ZOOPLANKTON							
			MACROINVERTEBRATES & FISH							
			WATER QUALITY							
MONTICELLO SUB-IMPOUNDMENT STATION	L	12	PHYTOPLANKTON							
			ZOOPLANKTON							
			MACROINVERTEBRATES & FISH							
			WATER QUALITY							
	M	13	PHYTOPLANKTON							
			ZOOPLANKTON							
			MACROINVERTEBRATES & FISH							
			WATER QUALITY							
	N	14	PHYTOPLANKTON							
			ZOOPLANKTON							
			MACROINVERTEBRATES & FISH							
			WATER QUALITY							
NEAL SHOALS DAM STATION	O	20	PHYTOPLANKTON							
			ZOOPLANKTON							
			MACROINVERTEBRATES & FISH							
			WATER QUALITY							
	H	18	PHYTOPLANKTON							
			ZOOPLANKTON							
			MACROINVERTEBRATES & FISH							
			WATER QUALITY							
	P	11	PHYTOPLANKTON							
			ZOOPLANKTON							
			MACROINVERTEBRATES & FISH							
			WATER QUALITY							
BROAD RIVER STATIONS	SA		WATER QUALITY							
			WATER QUALITY							
			WATER QUALITY							
			WATER QUALITY							
			WATER QUALITY							
			WATER QUALITY							
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			WATER QUALITY							
			WATER QUALITY							
			WATER QUALITY							
			WATER QUALITY							

Figure 3. Summary of Biological Monitoring Requirements in 1978.

VIRGIL C. SUMNER NUCLEAR STATION PROJECT

			1983-84											
			JAN.	FEB.	MAR.	APR.	MAY	JUNE	JULY	AUG.	SEPT.	OCT.	NOV.	DEC.
LOCATION	BIOLOGICAL STATION	WATER QUALITY STATION	INTEGRATED AERIAL PHOTOGRAPHY (1/23/83 - 1/27/84)											
			WATER FROM SURVEY (IN) 1/23/83 - 1/27/84											
SAMPLE TYPE			SURVEY MAP AND IDENTIFY VASCULAR HYPOPHYTES ID IN PARR AND MONTICELLO RESERVOIRS											
REQUIRED BY			SURVEY MAP AND IDENTIFY VASCULAR HYPOPHYTES ID IN PARR AND MONTICELLO RESERVOIRS											
D - DITCH			SURVEY MAP AND IDENTIFY VASCULAR HYPOPHYTES ID IN PARR AND MONTICELLO RESERVOIRS											
N - NUC			SURVEY MAP AND IDENTIFY VASCULAR HYPOPHYTES ID IN PARR AND MONTICELLO RESERVOIRS											
1/4 QUARTERLY SAMPLE			1/4 QUARTERLY SAMPLE											
1/2 QUARTERLY SAMPLE			1/2 QUARTERLY SAMPLE											
3/4 QUARTERLY SAMPLE			3/4 QUARTERLY SAMPLE											
1 QUARTERLY SAMPLE			1 QUARTERLY SAMPLE											
PARR RESERVOIR STATIONS			PARR RESERVOIR STATIONS											
B 2			B 2											
C 2W			C 2W											
D			D											
MONTICELLO RESERVOIR STATIONS			MONTICELLO RESERVOIR STATIONS											
I 17			I 17											
J 16			J 16											
K 15			K 15											
L 12			L 12											
M 13			M 13											
N 14			N 14											
O 20			O 20											
MONTICELLO SUB-IMPONDUENT STATION			MONTICELLO SUB-IMPONDUENT STATION											
H 18			H 18											
NEAL SHOALS DAM STATION			NEAL SHOALS DAM STATION											
P 11			P 11											
BROAD RIVER STATIONS			BROAD RIVER STATIONS											
SA			SA											
I			I											
			JAN.	FEB.	MAR.	APR.	MAY	JUNE	JULY	AUG.	SEPT.	OCT.	NOV.	DEC.

Figure 4. Summary of Biological Monitoring Requirement in 1983-84.

dirt from within the reservoir area. In addition to these measures a land management program was instituted to speed the ecological adjustment of the project area and further alleviate the adverse effects of construction. This program included planting ground cover for erosion control, multiple use timber management, vector control, and landscaping. Other mitigating measures included wildlife, fishery, and recreational management programs. The fishery and wildlife management programs were closely associated with the biological studies. Those studies helped plan effective mitigation programs which combined with the recreation management program produced a diverse, popular, public use program.

In order to mitigate the loss of wildlife habitat caused by the construction of the power complex a decision was made to utilize the newly created lake habitat to its fullest potential. A 320-acre, constant level subimpoundment was created in the northern end of Monticello Reservoir for the purpose of recreational fishing, swimming, and picnicking. The subimpoundment was intensively managed as a recreational fishery by a fish stocking program, and nutrient enrichment program.

On Monticello Reservoir boat ramps and picnic areas were created to stimulate public usage and to provide access to the rapidly developing natural fishery typical in a young lake. At one of the picnic areas the recreation potential was upgraded by building public tennis courts and a softball field.

At Parr Reservoir several enhancement features improved the area for public use. These included:

- ° Addition of several boat launching ramps and camping facilities.
- ° Initiation of a waterfowl management plan.

The water fowl management plan included the development of a dike-controlled marsh area with adjacent fields. Both the marshes and the fields were planted with preferred wildlife foods and cover vegetation suitable for nesting. Wood duck nesting boxes were installed in suitable backwater locations. The waterfowl management plan also included the development of a green-tree reservoir near Parr Reservoir. The water level of this green-tree reservoir could be raised or lowered in order to enhance growth and availability of waterfowl foods.

Additional mitigation was provided by the initiation of a Canada Goose nesting and stocking program designed to establish a native Canada goose population in the Piedmont of South Carolina. This effort included transplanting geese from outside the state, establishment of food plots and nesting habitat development on the islands in Monticello Reservoir. To date, all of the mitigation programs are considered to be highly successful.

To ensure continued spawning success of the fishes of the Broad River downstream of the project area, the Parr hydroelectric facility maintained certain minimum downstream flow requirements.

SUMMARY

South Carolina Electric & Gas Company planned, constructed, and licensed a major power generating complex. Included in the complex were a newly constructed major pumped storage facility and a nuclear facility; an existing hydroelectric facility was updated.

Extensive planning and coordination with regulatory agencies was carried out to assess potential impacts on natural resources. A monitoring program was established in the project area, baseline conditions were established, and monitoring requirements were met during the construction period. A five-year post-operational monitoring requirement was met for the Federal Energy Regulatory Commission and a 2-year post-operational monitoring requirement was fulfilled for the Nuclear Regulatory Commission and the South Carolina Department of Health and Environmental Control.

Extensive mitigation measures were employed throughout the area encompassed by the power complex. During the construction period, erosion and dust control measures were used extensively. A land management program was instituted to hasten ecological adjustment. A timber management program for multiple use purposes was developed. Wildlife habitat improvement practices were established. Fisheries management programs including stocking of game fish for recreation purposes were also established. Other public use mitigation measures such as the establishment of tennis courts, ball fields, picnic areas, and a swimming beach were put into place.

This project was well planned and designed. The objectives of the owner and the regulatory agencies were fulfilled.

ACKNOWLEDGEMENTS

This work was supported by South Carolina Electric & Gas Company as part of the regulatory requirements stipulated in the articles to the licenses for each of the generating facilities. Special recognition is given to Messrs. W.E. Moore and W.R. Baehr of South Carolina Electric & Gas Company for their contribution to the overall success of the program.

IMPACT MITIGATION AND ENVIRONMENTAL ENHANCEMENT

REVIEW PAPER

John Christian
U.S. Fish and Wildlife Service
75 Spring Street Southwest
Atlanta, Georgia 30303

After reviewing the papers from the session, "Impact Mitigation and Environmental Enhancement", I feel the key issues are: 1. what is the ultimate impact on the resource of importance, and 2. what is the best way to design a program to answer that question. If we cannot answer that question, can we develop an iterative dynamic process so that we can make changes along the way?

All the papers presented in my session offer good examples of the application of various principles of impact mitigation and environmental enhancement. My objectives are to explain the overall principles and illustrate how the specific papers serve these principles.

The necessary starting point is to understand what is meant by the term environmental impact. The definition and use of this term is a critical one for all of us. If there is no environmental impact, then there is no need to consider mitigation, but as I will explain in more detail later, enhancement certainly could continue to be a project planning objective. The dictionary defines the word "impact" as a concentrated force producing change. This definition does not convey any meaning that an impact is necessarily bad or good. It merely conveys the meaning that an impact occurs when the existing condition is changed. Any impact, then, can be interpreted as good or bad, depending on your point of view and your point of reference.

Ross Wilcox has given us some excellent examples of what I mean from the standpoint of the manatees' reliance on thermal plumes at Florida Power and Light coastal power plants. In this case, thermal effluent is a good impact and provides a benefit to an endangered species. However, a strict preservationist might consider such an artificially created symbiotic relationship as a negative impact because it does not represent the natural situation. Again, it depends on your point of view.

Likewise, a marine biologist who specializes in plankton communities may consider a thermal plume as having an adverse impact on the nearshore plankton community. But that same thermal plume may serve to attract a fish species of sport value, and the fisherman is likely to have a different point of view than the marine biologist as to whether this impact is bad or good. However, if it could be shown that the species composition shifts would result in a reduction of fish productivity in the overall area, perhaps both the marine biologist and the fisherman would agree that the impact is adverse.

Ecologists might disagree with my comments and say that the primary objective is maintenance of the structure and function of the natural ecosystem and that an adverse impact occurs when that natural ecosystem is disrupted. My response is that I do not believe that there are many remaining examples of natural ecosystems that man has not altered or impacted. There may be some examples in the most wild of the wilderness areas, but we generally do not build power plants in those areas. Any ecologist or biologist knows that the structure and function of ecosystems are dynamic and change over time. Microhabitats are created and then destroyed through natural processes. The overall integrity of the larger system is certainly buttressed against major change and provides the conditions that make that particular ecosystem or habitat type unique in terms of the plants and animals that live there.

In the majority of cases, development projects do not affect entire ecosystems. Individual projects always have the capability to cause major changes in local populations of plants and animals. However, the cumulative effect of many such projects certainly has the capability to provide a major system disruption. The point that I am trying to make is that the word impact is a relative term with some subjective components. An impact denotes change; change can be good or bad, depending on your point of view. A clear understanding of whether an impact is considered adverse is a prerequisite to mitigation planning.

Robert Davis' paper on angled intake screens provides an example. In his presentation, he indicated that they were very successful at mitigating impacts for certain species; specifically, one example was the winter flounder. There was very little loss to the flounder, both in

terms of initial impact or long term impact. However, for the bay anchovy, it was a failure. The angled screen did very little to avoid mortality of the bay anchovy. The question is what is the relative ecological significance and importance of those species. One needs to answer that to make a determination about whether that particular mitigation measure is going to result in a good impact or a negative impact.

An initial resource inventory that identifies critical biological areas within the zone of influence of the project and the primary biological resources that might be affected can serve as a starting point for the impact analysis. Impacts should be defined by a multidisciplinary team that includes fishery biologists, wildlife biologists, system ecologists, limnologists or marine biologists, depending upon where you are, and recreational users, such as fishermen, hunters, and recreational boaters. The result should be a list of environmental objectives for the area in question from which conclusions can be drawn about whether a change caused by a given power plant design would result or could result in an adverse impact, a beneficial impact, or no impact at all, based on the agreed upon environmental objectives.

Linda Cadman's paper I think provides another example of this kind of thinking and philosophy. The objective of lowering the Δt through the use of tempering pumps had a very severe negative impact through entrainment of important fish species. So the objective of lowering the t through tempering pumps perhaps was the wrong objective to have set. Now, as I understand the case that Linda presented, there was no way to anticipate the relative effects prior to construction. There may have been data that could have been collected, experiments run, but my perception was there was no way to identify that initially, and that there had to be a dynamic response including continued monitoring, to identify that effect, and in essence, reevaluate that objective of lowering the Δt . Mitigation only needs to be considered for those impacts determined to be adverse in nature.

I will now outline the principles of mitigation from the standpoint of the Fish and Wildlife Service. The Service has for many years been involved in reviewing projects and providing mitigation recommendations

to agencies or private developers to mitigate those impacts the Service feels are adverse. I have some slides which illustrate what I am talking about. A formal mitigation policy was developed in 1981 which incorporated all the principles and concepts that the Service has used throughout the years and that each of you have used in your project planning. Each of the speaker's papers illustrates one or more of the mitigation principles that are embodied in the Fish and Wildlife Service policy. I will review the definition of mitigation, as outlined by Dr. Jim Brown and his paper on mitigation banking. The President's Council on Environmental Quality defines mitigation as a series of steps to be taken to eliminate an adverse impact. As Dr. Brown indicated, these items are not a menu of equivalent options for mitigation. Recognizing the definition as a sequence of mitigation steps that begin during the initial planning stage of the project will help speed the approval of the project and reduce the need for expensive modifications later on in the regulatory process.

Ross Wilcox's paper outlined principles that address all of the definitions of mitigation. I will address Ross' efforts later because they are very creative; they are innovative and different from application of the term mitigation. Linda Cadman's paper on the use of tempering pumps relates to minimizing impact and rectifying the impact over time. There was an effect that was determined later on; that effect was subsequently reduced or eliminated over time. Robert Davis' paper on angled screens focused on minimizing impact. Roger Rulifson's paper focused on the first step -- avoiding the impact. Roger's paper reflects a possible future for power generation that, as biologists and project managers, we need to think about so that we can understand and develop ways to avoid an impact for the mutual benefit of both the company and the agencies. Dr. Jim Brown's presentation focused on ways to compensate for unavoidable losses. Charles Zimmerman's paper addressed all the principles in a formal sense of a major planning project for a power generation complex that was extremely successful.

The Fish and Wildlife Service mitigation process covers all federal projects and private projects that require a federal permit or license. The Fish and Wildlife Service reviews and recommends actions; the agency

or the private project developer then implements those actions. The purposes of the policy are to: 1) assure consistent and effective recommendations that will conserve important fish and wildlife; 2) to facilitate balanced development and multiple use of natural resources; 3) to allow other agencies and developers to anticipate Service recommendations; 4) to reduce delays and conflicts; and 5) to clarify that it is not an acre-for-acre policy. This policy can apply to both habitat losses as well as losses of individuals or population impacts.

The preferred approach to mitigation is to provide information in a form and in time to avoid or minimize fish and wildlife losses as part of the initial project design. Roger Rulifson attempted to do that for us. He provided information for understanding and developing ways to mitigate those impacts. Charles Zimmerman gave us a more recent example of applying the principle to a major power complex successfully. It is clear to me that Mr. Zimmerman identified environmental objectives early, developed an information base, determined what each of the agencies' requirements were, added enhancement features, and in a very cooperative way undertook a very successful project.

The first major aspect of the Fish and Wildlife Service mitigation policy is to establish mitigation goals by resource category (Table 1). As I indicated earlier, not all impacts are adverse and, in some cases, some impacts are more adverse than others. The Service has recognized this by establishing a number of resource categories that include designation criteria and mitigation goals. Please note that, depending on the value and scarcity of the resource involved, the mitigation goal ranges from complete preservation of the area to no compensation required at all, only efforts to minimize loss. The establishment of these mitigation goals early in the process is important. Mitigation goals are essential and assist in developing the resource objectives. The resource objectives will, in fact, be used to apply the criteria of the mitigation policy to those species that are selected. We recommend avoidance or full compensation for the most valued resources and that the degree of mitigation correspond to the importance and scarcity of the resource.

The determination of importance is based on BPJ technology, that is, the Best Professional Judgment of the involved biologists. If you think

Table 1. Mitigation Policy.

RESOURCE CATEGORY	HABITAT CRITERIA	MITIGATION GOAL
1	HIGH VALUE IRREPLACEABLE	NO LOSS
2	HIGH VALUE SCARCE	IN-KIND
3	HIGH TO MEDIUM ABUNDANT	NO NET LOSS
4	MEDIUM TO LOW	MINIMIZE LOSS

about that, you will no doubt conclude that majority of environmental decisions today involve BPJ as opposed to BPT or BAT. Now, in terms of this policy, I will explain the mitigation goal. In resource category 1, because that habitat area is irreplaceable, limited and of extremely high value, the biological principle is that, since it cannot be replaced, we will not tolerate any loss. Resource category 2 has high value for evaluation species and the habitat itself is scarce. However, it can be compensated for, through creation of additional habitat of that same kind. So in-kind compensation is the mitigation goal for resource category 2 habitat.

An example of a resource category 2 habitat might be a bottomland hardwood swamp. We cannot simply say that bottomland hardwood swamps have to be mitigated and compensated for in-kind. The reason is that not all bottomland hardwood swamps are equally valuable to wildlife. For instance, a five acre bottomland hardwood swamp area in Louisiana we called a resource category 4 because, in that situation, it had a low species value. The bottomland hardwood swamps of the Atchafalaya River, though, have tremendous wildlife value, and we would designate those bottomland hardwood swamps as a resource category 2 and strive to achieve no net loss and in-kind compensation for the losses of bottomland hardwood habitats.

The Service has developed a number of impact assessment methodologies. The primary guideline is to consider the impact in terms of the difference between the future with the project and the future without the project. The Service seeks to evaluate habitat value in biological terms instead of economic terms and has developed a specific methodology to do so. That methodology is based on subjective judgment, in part. It is also based on data analysis and interpretation, and if any of you are interested, I have a Habitat Evaluation Procedures Suitability Index for the bald eagle that you can look at so you can get a feel for how that process works. But once you select your species, you can determine the relative value of that habitat based on the life history and habitat requirements of a particular animal. This can be quantified and used in developing an objective analysis of what is necessary to mitigate losses for that particular animal. In the

endangered species area, we have not yet developed a specific methodology that deals with direct impacts to the animals. The habitat evaluation procedures will apply to endangered species, as I just indicated in the bald eagle example.

In the case of entrainment or impingement where you are having impacts on the animals themselves with no direct impact to the habitat, there are no specific methodologies that have been developed for quantifying or developing a mitigation strategy. Instead, we develop mitigation strategies based on our best professional judgement. We try to remain as flexible as possible; I do not want to leave you with the impression that we have a mathematical formula which spits out a mitigation plan, but we do have both quantitative and qualitative methods for determining a mitigation plan for a given area.

The Service also requires that its mitigation recommendations be cost effective in terms of achieving the mitigation goal. We ask our biologists to consider the economic cost of recommended strategies and to recommend the least costly one. We also encourage the project sponsor to incorporate funding for mitigation as a legitimate project cost and to carry out any compensation plan at the nearest site possible within the project area. Although, given our flexible policy, it is possible for the project sponsor to mitigate and compensate for resources lost off site. Now I will give you some examples of how the Service implements its policy.

The first example is a very simple set of situations that could apply to the siting of a major power generating facility. Application of the Service's mitigation policy would result in the Fish and Wildlife Service strongly opposing the siting of the plant at site number 1 because of thermal impact on the major blue crab wintering area offshore from the proposed outfall at plant site 1. Please note that the power plant facility is located in a resource category 4 area, which has little wildlife habitat value. That does not mean that all upland habitats are resource category 4. A specific example related to the endangered red-cockaded woodpecker; it requires mature pine forest habitat, upland habitat, old growth timber, and in that case, that habitat would be a resource category 1. So, it depends on what is there, what species are

selected and what your objectives are in terms of environmental goals for that area.

Site number 2 is located in a resource category 2 wetlands with an outfall into the deep ocean, which has been designated as resource category 4.

No mitigation is required for the ocean outfall. However, the wetland impact requires that there be avoidance and minimization as well as replacement of lost habitat because the goal for that particular resource category, is no net loss of habitat value. The cross hatched area could be scraped down and converted into a wetland to offset the loss as a result of the acreage taken out by the power plant at site 2.

Site number 3 is located in resource category 4 habitat. In this case, the only requirement is to minimize the footprint of the project site. The outfall again discharges into a deep ocean area that we have determined has no fishery value that would be impacted by the outfall. This information can be invaluable to project planners and power plant management decision makers. Given a choice of sites, all else being equal, the manager would choose site 3 to build the plant. However, economics or land availability or some other factor may dictate that the power plant be sited at site 2. If it is, the additional adverse impact on the public resource can be accommodated through compensation of the habitat loss. However, if the plant manager's only option is site number 1, break out the lawyers, because you are in for a fight. We have a good policy and a strong biological rationale to stop your project and require a closed cooling system.

Next, I will discuss the concept of environmental values. As used by the Fish and Wildlife Service, the term enhancement refers to developing or improving fish and wildlife values beyond that which would exist without a project. None of the legal authorities that govern the Fish and Wildlife Service or other agencies require that fish and wildlife values be enhanced during economic development projects. However, the Fish and Wildlife Coordination Act does require that enhancement opportunities be identified during project planning by the Fish and Wildlife Service.

Enhancement actions by a power company have three primary benefits. The first is improved public relations. Projects such as those undertaken by Florida Power and Light Company have created tremendous positive feelings on the part of the public and the Fish and Wildlife Service toward this company. Charles Zimmerman's power complex example has no doubt generated similar positive feelings in the local community among the agencies involved. This improves future communication and resolution of conflicts that might arise later on.

The second benefit is that the enhancement action can benefit the company economically through the mitigation banking concept. The third major benefit to the company is the altruistic one from the standpoint of making things better than they were before. I have been told that there are a number of chief executives that have been known to engage in this type of altruistic thinking, particularly if it creates a tax advantage.

Figure 1 summarizes the concepts of mitigation and enhancement. The vertical axis represents the environmental condition of the project site. The point labeled zero is the existing condition without the project in place. The scale measures the degree and characterization of impact as a result of the project, and it's labeled "better or worse". The scale indicates that the environmental conditions could improve or could be worse. The horizontal scale represents units of time starting from zero, which represent the point at which the project creates its impact running through time ten. The first horizontal line represents a worst case environmental impact for a project with poor planning and no mitigation at all. Please note that the area of impact is proportionately large and continues throughout time forever, that is, without any form of mitigation. The cumulative effect of this type of project can certainly result in a breakdown in the structure and function of the aquatic or terrestrial ecosystem and subsequent loss of important fish and wildlife species and other important environmental functions. That is one of the major problems that we are dealing with right now in terms of manatee protection. We have docks being constructed which have a very small incremental impact that we cannot conclude would jeopardize the continued existence of the manatee, but nonetheless as those small incremental impacts add up, it certainly will jeopardize the continued existence of

the manatee. Docks do not kill manatees, but the boats that use the docks do. Not all the boats, certainly.

The second horizontal line represents a project that had good environmental planning but no compensation of resources lost. Note that the area of impact has been minimized. The third horizontal line represents a project with no impact at all. The diagonal line labeled project number 1 represents a project with no adverse environmental impact for which enhancement features have been added. It turns out that Florida Power and Light power plants with their thermal discharges have generated no impact. The concern in terms of the negotiations with the company years ago was that it may well create an adverse impact. A mitigation plan was developed based on that potential for adverse impact. As it turned out, it was not an adverse impact. So theoretically, anyway, those actions would be considered enhancement actions. The area to the right of the line represents environmental benefits that will accrue through time and for which the project sponsor could claim mitigation banking credit to offset future project losses. The line labeled project 2 represents a project with good environmental planning that has applied all aspects of mitigation and enhancement. The project has been designed to minimize impacts, unavoidable impacts have been compensated, and enhancement actions have also been designed into the project to create mitigation banking credit. The diagonal line labeled project number 3 represents a poorly planned project with a severe environmental impact for which full compensation was required. Please note that the area of impact to the left of the line is significantly larger than the area of impact for the well designed project. And at an economic level it may have been more cost effective for the project's sponsor to pay for good planning than to have to pay the additional incremental cost for mitigating the environmental resource damage.

I certainly hope that this puts the papers presented during my session into perspective as examples of the various kinds of approaches for mitigation and enhancement. Creativity, innovation and trust are the key.

I would like to thank my speakers for the illustrations they provided and offer the following summary comments.

1. Power generation planning should include environmental objective setting as well as economic and regulatory objective setting.

2. Environmental objectives have the largest subjective component. They should be developed through a process of best professional judgment and with public input.

3. An adverse environmental impact only occurs when there is a difference between the desired condition and the actual condition as a result of the project. Environmental objectives must be defined as a prerequisite to defining adverse environmental impacts.

4. Adverse environmental impacts can be avoided or minimized through good environmental planning at the earliest point in the project planning process.

5. Mitigation measures should be evaluated in terms of ecological cost as well as the ecological benefits.

6. Mitigation measures need to be maintained and reevaluated at periodic intervals to evaluate performance.

7. Unavoidable adverse environmental impacts can be compensated for by replacing or substituting those resources lost by the action.

8. Environmental tradeoffs may be necessary to compensate for resources lost; for example, creation of wetlands in return for loss of shallow water habitat.

9. Positive environmental impacts can result from power generation activities.

10. Enhancement action can build public and agency support for existing and future power generation activities and create economic values through creation of mitigation banks to reduce future costs of regulatory compliance.

11. The process requires open communication, a desire on the part of all parties to accommodate each other's objectives, and a flexible regulatory setting within which to apply these principles. If it is done right, then everyone wins.

I am going to close with one short example of a recent situation that the Fish and Wildlife Service was involved in. It involved the L reactor which is operated at the Savannah River Plant. They needed to develop a cooling system and proposed to build a cooling lake. The impact of the

cooling lake would have resulted in a regulated flow that would have inundated Steel Creek. Steel Creek was used as a foraging area for the endangered wood stork and the Birdsville Colony, which is located approximately 30 miles from the plant site, contains our most northernmost population of the wood stork, with over 100 birds. We determined that Steel Creek is a foraging habitat for that particular population. Early in the process, we worked with the Department of Energy to develop a mitigation plan to offset the loss of Steel Creek. It became highly charged emotionally and politically and, as you can imagine with anything that relates to nuclear energy or weapons development, everyone was offering opinions. A typical example was a Scrawls cartoon in the Atlanta Journal Constitution showing three witches stirring a pot. The pot was labeled "the L reactor cooling lake" and the caption was "bubble, bubble, toil and trouble. Cook the woodstork; fry his brain; another of God's creatures down the drain."

Working in that highly charged atmosphere with the DOE and their personnel, we invited Audubon to assist us and we developed a mitigation plan which created alternate foraging habitat on Audubon land located north of the project site. That project cost DOE approximately half a million dollars to construct and involved the creation of a reservoir designed specifically to provide foraging habitat for the wood stork. Because they are tactile feeders with their bills, wood storks require specific water levels and size fish to make a living. The reservoir was designed specifically for this type of foraging habitat. Many people did not think it would work and it involved a risk on our part, the Audubon's part, and the Department of Energy's part. The good news is that this summer over half of the Birdsville population of wood storks made extensive use of the Kathwood Lake mitigation site. The biologists feel that it was a tremendous success and will provide a more stable source of feeding area for the wood stork than did Steel Creek, which was subject to natural water fluctuations.

I hope these review comments and my speaker's papers will help to stimulate such a win-win scenario in power generation planning.

SOME OBSERVATIONS ON AQUATIC IMPACTS OF
POWER PLANT TRACE ELEMENT DISCHARGES

Peter M. Cumbie
Design Engineering Department
Duke Power Company
P.O. Box 33189
Charlotte, North Carolina 28242

ABSTRACT

Trace elements discharged from fossil-fueled steam electric power plants are frequently a source of concern in relation to possible toxic effects on aquatic life or exceedances of water quality standards in receiving waters. This paper discusses several aspects of trace element discharges and their effects on aquatic ecosystems.

Factors which affect the potential for toxic effects include the mass of pollutant discharged, the rate of flow in the receiving water body, potential for bioaccumulation or biomagnification, interactions with chemical factors, temperature, presence of other toxics, and accumulation in sediments.

Practical examples of the effects of these and other factors which may lead to no effect, limited effect, or severe effect on aquatic organisms are presented. Implications of observed effects for some widely-held assumptions concerning responses of aquatic life to toxic pollutants are discussed.

INTRODUCTION

The purpose of this paper is to share experiences with a trace element discharge problem. Although somewhat obvious when considered in retrospect, some of the problems described here were not readily interpretable upon first consideration. The case described here concerns the effects of selenium and arsenic discharges on fish and other aquatic organisms in Belews Lake, North Carolina.

Belews Lake is a 1563-ha power plant cooling lake located near the city of Winston-Salem. Belews Lake received effluent from a fly ash settling basin at the Duke Power Belews Creek Steam Station from 1975 through 1984. After consideration of various alternatives, a dry fly ash collection system was installed at the plant in 1984. This system eliminated wet sluicing of fly ash and discharge of fly ash sluice water from the settling basin to Belews Lake (Cumbie et al, 1985).

In 1976, less than two years after fly ash settling pond effluent began to enter Belews Lake, fish in the reservoir failed to reproduce and began to suffer excessive mortality, based on results of electrofishing, rotenone sampling, and observations of dead fish. All species were affected, but Centrarchids seemed to be most sensitive. Studies eventually showed that the discharge of selenium to the reservoir, with subsequent bioaccumulation by plankton, benthic organisms, and fish was the cause of these problems (Cumbie and VanHorn, 1978). Recent effects of selenium in the Kesterson Reservoir in California have brought national attention to this element as a subject of concern in aquatic ecosystems (Saiki, 1985). Selenium and other trace elements have accumulated in the Kesterson system from agricultural drainage in the SAn Joaquin Valley, resulting in mortality of fish and bird life. But in 1976, the potential for selenium to cause ecological problems in aquatic systems was not generally recognized.

The Belews Lake selenium problem has been described in a number of reports and technical papers (Cumbie and VanHorn, 1978; Cumbie, 1978; Cumbie et al, 1985), and the details of these studies will not be presented here. Instead, this paper discusses a number of issues with application to trace element discharges in other water bodies. It is hoped that these observations will stimulate further research and

heightened appreciation of factors contributing to trace element effects on aquatic life.

THE DISCHARGE AND THE SYSTEM

Initially, the nature of the receiving water system and its relation to trace element effects may be considered. This subject is discussed first because it has implications for many types of discharges, not just for trace elements or for any particular type of chemical compound. Belews Lake is unusual because of its lengthy retention time of approximately 1500 days or about 4.5 years when the combined effects of limited natural tributary streamflow and evaporative losses due to the power plant heat load are considered (Cumbie, 1978; Weiss and Anderson, 1978). It is obvious that discharges of a conservative pollutant to such a system would have a great potential for toxic effect, especially if the quantity discharged significantly exceeded the natural inputs of the substance. The design of the Belews Creek Steam Station called for wet sluicing of fly ash to an ash settling basin, from which the sluice water was discharged back to Belews Lake because of the quantity of water involved and the need to maintain the water level in the reservoir. It was also desired to make the project self-contained in a regulatory sense, with the alternative being to discharge ash basin effluent to the nearby Dan River.

Although the potential for problems with a discharge of conservative pollutants to a system of great retention time seems obvious, it has not over the past several years prevented outside observers from questioning why, if selenium discharges caused serious problems with fish populations in Belews Lake, ostensibly similar discharges did not cause the same problems in other receiving waters. It has developed that other systems receiving selenium discharges without associated deleterious biological effects differed from Belews Lake in terms of a much shorter retention time, greater productivity or sedimentation rates, or in other important respects. This presents something of a paradox: effects of selenium discharged to Belews Lake were a product of a particular combination of factors including retention time, general water quality, biological productivity, chemical properties of coal and fly ash, and the quantity of discharge (Cumbie, 1978, 1980). Still, other systems may share a

sufficient number of these properties that the problems experienced in Belews Lake can offer useful guidance to interpretation of effects of other trace element discharges.

CONTINUOUS DISCHARGES VERSUS SPILLS

The Belews Lake ash basin effluent was a continuous discharge which occurred as a recognized operating feature of the power plant (Cumbie, 1978; Cumbie et al, 1985). Although the introduction of selenium to Belews Lake did not cause immediate responses in the form of a fish kill or violations of numerical water quality standards, the continued discharge eventually resulted in a sufficient loading of selenium in the system to cause extermination of most fish in the main basin of the reservoir. The Kesterson Reservoir case is similar in the sense that introduction of selenium to the reservoir and associated evaporation ponds from agricultural drainage has been continuous (Saiki, 1985).

Large, one-time releases of toxic substances to receiving water have caused more spectacular results in the form of fish kills, but such problems have generally proved to be short-lived. In these cases, acutely toxic conditions are developed, but are rapidly abated by the combined effects of chemical decay, precipitation, or dilution of the responsible agent. Methods for evaluating continuous discharge effects must obviously be different from those which are used for spills. The latter often attract immediate attention from operating personnel, regulators, and the general public. In contrast, more insidious effects of continuous discharges may go largely unnoticed, even when highly significant ecological responses are occurring. As discussed below, routine monitoring programs may not provide reliable indications of these effects as they develop.

MULTI-ELEMENT DISCHARGES

It may be difficult to separate the effects of various trace elements which are present in a discharge. The impacts of selenium in Belews Lake have been difficult to separate clearly from possible effects of arsenic, which was also present in the ash basin effluent. The presence of arsenic in the Belews Lake system has tended to draw more attention from outside observers. This is apparently due to greater familiarity with

arsenic as a toxicant, compared to selenium. In fact, the levels of arsenic in Belews Lake water did not exceed water quality standards, and levels accumulated in sediments have not been sufficiently high to lead to predictions of direct effects (Cumbie, 1978; U.S. EPA, 1979). The situation with selenium differed, since available data indicated that the quantities of selenium alone taken up by fish in Belews Lake were sufficient to cause toxic effects and mortality, without additional effects of any other agents that might be associated with selenium or otherwise present (Goettl and Davies, 1976). Recent experimental data obtained at Carolina Power and Light Company (S.E. Woock, personal communication) indicate that selenium alone can account for virtually all of the adverse effects on fish reproduction which have occurred in Belews Lake.

Synergistic or additive effects of selenium, arsenic or other substances on fish populations in Belews Lake must be considered. However, water quality criteria for constituents other than selenium, including arsenic, were not exceeded in Belews Lake (Cumbie, 1978; Weiss and Anderson, 1978). although synergistic or additive effects of toxic pollutants may be important in some cases, scant available data suggest that these effects occur with acutely toxic exposure concentrations, rather than with chronically toxic concentrations (Spehar and Fiandt, 1985). If a single element is documented to be capable of causing an observed effect, treatment of the identified problem is appropriate until additional evidence implicates other causal factors. A theory or hypothesis dealing with toxic effect should be made no more complex than necessary to explain the phenomena of interest. Other cases will undoubtedly be encountered in which adverse effects of particular agents can be recognized, even when other toxicants are suspected to be present. A focus on evaluation of synergistic effects would often not be an efficient approach to a solution for such a problem.

WATER QUALITY AND STANDARDS

Water quality criteria for selenium and arsenic which were available in 1976 would not predict adverse effects of these elements in Belews Lake. EPA "Red Book" criteria for selenium (U.S. EPA, 1976) at 10 ug/l suggested that lake water selenium levels of 8 to 12 ug/l would not lead

to problems with fish populations. This caused attention to shift temporarily to other possible causes for fish population declines, such as diseases, pesticides, adverse physical conditions, excess heat, or water level fluctuations. All of these were eventually eliminated as causes of the problem. As recently as 1982, the EPA national water quality criteria for selenium were set at 35 ug/l (U.S. EPA, 1980). This criterion is greatly in excess of the selenium concentrations detected in Belews Lake at the time that extremely high bioaccumulation of selenium was occurring in fish. Levels of 30 to 50 ppm (wet weight) were detected in fish muscle tissue, compared to normal values of about 1 ppm. This phenomenon was associated with widespread mortality of many species, and a more-or-less complete failure of reproduction among all species (Cumbie and Van Horn, 1978).

The problem with application of water quality criteria stemmed from the fact that the observed effects were not caused by direct exposure to selenium in the water column, but were instead related to sediment and food chain contamination (Cumbie, 1978; Finley, 1985). Water quality criteria did not adequately deal with this situation. The EPA national criteria only consider bioaccumulation as it affects human consumption of fish and shellfish. Criteria are not developed based on direct effects of bioaccumulation in aquatic food chains. The State of North Carolina has revised its water quality standards for selenium to 5 ug/l in ponds, lakes, and reservoirs, based in part on effects seen in Belews Lake. EPA does not generally recognize a role for field observations in validating the accuracy of water quality criteria based on laboratory toxicity data, or in suggesting site-specific revisions to national criteria (U.S. EPA, 1986).

SENSITIVE SPECIES

Several interesting observations have been made regarding the apparent sensitivity of various species to presence of selenium in Belews Lake. These observations have applications for other cases.

It has been stated that benthic organisms exhibit greater sensitivity than other forms to polluted conditions because they are long-term residents in polluted waters and tend to integrate effects of pollutant exposure over time. This view is largely based on effects of elevated

temperature, low dissolved oxygen, or acutely toxic chemicals on stream benthos (Garton and Harkins, 1970). As fish populations declined in Belews Lake, extensive studies of benthic organisms revealed no recognizable abnormal changes in benthic biomass, diversity, or species presence (Weiss and Anderson, 1978). The only unusual features noted were a tendency for benthos to attain rather large body sizes and to be plentiful in appropriate habitats. This was apparently due to the absence of predations on these organisms as fish disappeared from the lake. Therefore, experience in Belews Lake indicates that benthos are not necessarily sensitive indicators of trace element pollution which may have severe effects on other fauna.

More recent data indicate that Daphnids tend to be the most sensitive forms in acute toxicant exposures (U.S. EPA, 1986). However, zooplankton data revealed no unusual changes in species or population levels in Belews Lake. Studies of zooplankton, periphyton, phytoplankton, and benthos undertaken to evaluate effects of temperature elevation judges conditions in the Belews Lake ecosystem to be normal for the region in 1976, at the very time when fish populations were declining (Weiss and Anderson, 1978). These results were in contrast to the commonly advanced position that one could discern the impacts of unsuspected pollutants at an early stage by conducting broadly-based monitoring studies of plankton, benthos, or other biota. Observations in Belews Lake emphasize that adverse effects on fish populations may occur unaccompanied by related observed effects at lower trophic levels. It was concluded that directed scientific investigation, rather than survey data, was required to achieve an understanding of the factors affecting fish populations in Belews Lake.

Finally, the fathead minnow (*Pimephales promelas*) appeared in Belews Lake as the native Centrarchids and Ictalurids were eliminated. It is interesting to note that the fathead minnow is commonly used as an appropriately sensitive test organism in studies of toxicity of trace metals or other substances in the laboratory (Peltier and Weber, 1985). It appears that in Belews Lake, the position of the fathead minnow in the trophic system, and its tendency to use relatively shallow, near-shore habitat, prevented it from being exposed to the more contaminated sediment areas and associated food organisms. Sediment data indicate

that selenium levels increased with increasing depth and distance from shore (Cumbie, 1984).

The possibility of species adaptation or acclimation to toxicant exposure can also have important implications for programs intended to deal with effects of toxics. In acute toxicity tests, green sunfish (*Lepomis cyaneellus*) from Belews Lake were significantly more resistant to toxic effects of sodium selenite than hatchery green sunfish which had not been previously exposed to selenium (Cumbie and Velte, 1986). The increase in selenium resistance was hypothesized to have resulted from the long-term exposure of the tested fish stocks to selenium in the food chain of Belews Lake. This suggested that survival of certain native fish species in the lake under the existing contaminated conditions could not necessarily be taken as an indication that conditions would allow survival of stocked hatchery fish intended to facilitate recovery of game fish populations.

SYSTEM LOADINGS AND ECOLOGICAL EFFECTS

The Belews Lake case illustrated that the concentration of a trace element such as selenium in an effluent may not in itself give an indication of the potential for effects in the receiving water system. On the other hand, a simple load calculation may give valuable insight when compared to natural loadings from tributaries. In Belews Lake, some simple calculations based on effluent selenium concentration, effluent flow rate, tributary inflows, and background selenium concentrations (see Cumbie, 1978, 1979; Cumbie et al, 1985) indicated that the selenium load from the Belews Creek ash basin effluent (100 to 200 ug/l total selenium, average flow rate about 0.6 m³/sec) to Belews Lake was more than 32 times the natural input. Ambient selenium levels in tributary streams were below the 1 ug/l detection limit for analysis. Thus, the relative increase in selenium mass loading was at least one to two orders of magnitude above natural inputs. In a system with a high retention time, an increase of this magnitude in the input of a conservative substance such as selenium should be expected to have a high probability of producing adverse effects. Specific effects would depend on potential for bioaccumulation and distribution among different ecosystem compartments. In Belews Lake, the increased input was rapidly reflected

in marked accumulation of selenium in the sediments of the reservoir and in all biological compartments which were examined.

LOOKING FOR THE INEXPLICABLE

Occurrences of unusual events or conditions should be suspected as indicators of developing problems, but recognition of the significant events and their interpretation may be very difficult. In Belews Lake, such an early event was a large die-off of blueback herring (*Alosa aestivalis*) which occurred in April 1986, preceding the general failure of fish reproduction during the following summer (Duke Power Company, 1976). This open-water species feeds on plankton, and it may have succumbed due to selenium poisoning in its food chain as selenium levels increased in the reservoir during early 1976. Zooplankton in Belews Lake were later found to contain up to 100 ppm or more (dry weight) of total selenium, many times the levels which have been shown to cause mortality in fish feeding experiments (Cumbie, 1978; Goettl and Davies, 1976). Seasonal dieoffs of Clupeids (threadfin shad, *Dorosoma petenense*) are relatively common in certain southeastern reservoirs due to low water temperatures in the winter months, but these usually occur in December to February, not in the spring when water temperatures are rising. The possible significance of a spring fish kill involving blueback herring was not recognized at the time.

Another condition noted in Belews Lake fish in 1976 was "blindness", a clouded lens condition, which was common in Lepomids in the main reservoir. The significance of this condition, a relatively non-specific response to a variety of agents, could not be determined, although it was widespread in the lake and did not seem to occur in fish in the less selenium-contaminated headwater areas of Belews Lake. Recent feeding experiments with selenium at Carolina Power and Light Company (S.E. Woock, personal communication) have shown that this clouded eye lens condition can be caused in the laboratory by selenium poisoning in bluegills (*Lepomis macrochirus*).

Finally, exophthalmus or "popeye", generalized edema and swelling, and hemorrhaging of blood vessels in the eye, gill, and tail regions is a reported symptom of selenium poisoning (Ellis et al, 1937; Sorensen et al, 1984). These conditions occurred commonly in fishes in Belews Lake in 1977-1978, and in combination with water chemistry data and marked

bioaccumulation of selenium in fish muscle tissue and internal organs led to the hypothesis that selenium poisoning was responsible for the decline in Belews Lake fish populations which began in 1976 (Cumbie and Van Horn, 1978).

IMPORTANCE OF SIMPLE PHYSICAL CHARACTERISTICS

Properties of various ecosystems may cause them to respond rather differently to similar pollutant discharges. In Belews Lake, the power plant full-load cooling water pumping rate of 33 m³/sec results in a complete circulation of the lake volume of 228 x 10³m³ in about 40 days.

This creates a strong circulation pattern about the main portion of the reservoir, and led to a rather uniform distribution of dissolved selenium in the main basin of Belews Lake (Cumbie, 1978). In contrast, a reservoir with less circulation might be expected to exhibit a pollutant concentration pattern in water or sediments in which more accumulation would be seen nearer the source of the discharge. Or, in a stream or "run of the river" impoundment, low retention time might limit areas of effect to downstream locations, or prevent significant accumulations from developing at all.

Belews Lake stratifies strongly during the summer and early fall, and during the period from 1976 to 1984 developed a strong oxygen deficit and reducing conditions in the hypolimnion by late summer. This resulted in an obvious pattern in selenium concentrations, with peak concentrations developing in the summer, which could be readily related to the stratification of the reservoir and the chemical properties of selenium compounds (Cumbie, 1978; Weiss and Anderson, 1978). The low oxidation-reduction potential developed in the Belews Lake hypolimnion was hypothesized to reduce selenium to selenite or to insoluble elemental selenium, which was deposited in lake sediments (Cumbie, 1978, 1984). This process was critical in developing the selenium-contaminated benthic food chain which contributed to fish mortality. In other systems, where development of a reducing hypolimnion may not occur, the deposition of selenium in sediments would be expected to be less pronounced.

In the main basin of Belews Lake, selenium and arsenic levels in sediments were high relative to background conditions (Cumbie, 1984). But in uplake areas, accumulation in sediment was much less because of limited intrusion of the selenium-bearing waters into the headwater

areas, and because of a much higher rate of sediment deposition due to erosion in the watershed.

The productivity of Belews Lake is quite low (Weiss and Anderson, 1978), and the rate of accumulation of organic detritus and sediment is low. Deposition of inorganic stream-borne sediment is also very limited in the main reservoir because of limited tributary inflow and removal of sediment in the headwater areas. Therefore, sediment selenium and arsenic reached higher levels in Belews Lake (30 to 50 ppm total selenium or arsenic in surficial sediments) than those which have appeared in other, more productive reservoirs which have been subject to selenium loading from power plants (Carolina Power and Light Company, 1981; Cumbie, 1984). The high sediment selenium levels in Belews Lake led to a marked uptake of selenium, but not of arsenic, in fish through the food chain, with attendant reduced reproduction and increased mortality.

An example of a different result is the case of Adair Run, a West Virginia stream which received ashpond effluent resulting in average instream concentrations of approximately 40 ug/l total selenium (Nicholson, 1982; Whitaker, 1982; Sepcht et al, 1984). In this system, benthic organisms and stream fishes were not affected by the selenium exposure. However, sediment in the stream accumulated only 1 to 2 ppm selenium, with most of the element apparently moving through the system in dissolved form. This case emphasizes the critically important effects of retention time, oxidation-reduction potential, and sediment deposition on the effects of certain trace element discharges, and the water column concentrations which may be tolerated in different ecosystems.

CHEMICAL FORMS SHOULD BE INVESTIGATED

Knowledge of the chemical forms of toxic elements present in aquatic systems can be decisive in enabling us to understand their effects on biota and distribution in the ecosystem. In Belews Lake, circumstantial evidence indicated that selenate was formed in the fly ash sluicing system at pH 9 to 10, facilitating dissolution and discharge of selenium from the ash settling basin. In the lake, selenate appears to have been converted to selenite, which could be adsorbed by particulates and settled to the sediments, or further reduced to the insoluble elemental form in the summer hypolimnion, again resulting in deposition to sediments (Cumbie, 1978, 1980, 1984). Selenite is more toxic to aquatic

life than selenate (U.S. EPA, 1980). Behavior of selenium in the San Luis Drain and Kesterson Reservoir systems may differ from that observed in Belews Lake if more alkaline conditions favor maintenance of selenium in the form of selenate (see Saiki, 1985). Our present understanding of redox transitions and kinetics of many chemical reactions of interest in trace element toxicity is rudimentary. This is a fruitful area for future research.

CONCLUSIONS

Trace element discharges present significant challenges to the environmental scientist or engineer faced with interpretation of responses of aquatic systems to such discharges. This discussion of experiences with selenium in Belews Lake has sought to illustrate a variety of factors which should be considered in order to determine whether a discharge is having significant effects in a particular ecosystem. Since different ecosystems respond differently to pollutant discharges, depending on their individual properties, a discharge with significant adverse effects in one location may have no discernible effect in another location. Further research on ecosystem responses to trace element loading is needed to enhance our ability to distinguish these possibilities.

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RADIOCESIUM KINETICS IN THE
YELLOW-BELLIED TURTLE (Pseudemys scripta)

Eric L. Peters

I. Lehr Brisbin, Jr.

Savannah River Ecology Laboratory
Aiken, South Carolina

ABSTRACT

The recent Chernobyl accident has demonstrated the potential for radionuclide contamination of aquatic ecosystems. Although the initial release into the environment may be localized, animals inhabiting such contaminated areas may eventually serve as vectors of radionuclide spread into uncontaminated regions. In long-lived animals, such as the yellow-bellied turtle (Pseudemys scripta), the animals themselves may also serve as a reservoir of such contaminants, accumulating body burdens of radionuclides and releasing them slowly into the environment over extended periods of time. Essential to the determination of the fate of radionuclides released into aquatic ecosystems is an understanding of the mobility and retention of these substances in resident biota. To date, radionuclide uptake rates have been calculated from radionuclide elimination rates in a wide variety of aquatic species using the model of Davis and Foster. We tested this model by comparing the uptake kinetics calculated from radiocesium elimination rates of turtles held both in a controlled environment chamber and in an outdoor experimental pond, with the actual radiocesium uptake rates of turtles introduced into a ¹³⁷Cs contaminated reservoir on the Department of Energy's Savannah River Plant near Aiken, South Carolina. There were no significant differences between the turtles' indoor and outdoor elimination rates, both groups showing similar biological half-times. The relationship between the uptake and elimination rates was less clear; a significant lag period of 1-2 weeks was observed before the onset of radiocesium uptake. This lag period was attributed to a presumed delay in the onset of a regular feeding rate, while the turtles became accustomed to locating food in an unfamiliar area. This is in contrast to the Davis-Foster model, which assumes a uniform rate of radioisotope intake. Our results suggest that estimates of radionuclide dynamics (i.e., time to maximum radionuclide contamination in an individual organism) may not be estimatable from laboratory elimination data alone.

POLLUTANT ENRICHMENT OF SEDIMENTS FROM COOLING WATER
DISCHARGE IN BAHIA DE GUAYANILLA, PUERTO RICO

Rafael Almodóvar-Ferrer
José Manuel López
Center for Energy and Environment Research
University of Puerto Rico
College Station
Mayagüez, PR 00708

ABSTRACT

A 1125 MW oil fired power plant has been discharging cooling water in Bahía de Guayanilla since 1958-1959. The heated effluent enters an enclosed cove connected to the eastern portion of the bay by a narrow entrance. The concentrations of Ni, Zn, Cu and petroleum hydrocarbons in 4 stations within the intake area and in 4 stations in the thermal cove were determined. The sediments of the central portion of the cove have average concentration that are 217% greater for Ni, 69% greater for Zn, 62% greater for Cu, and 188% greater for petroleum hydrocarbons than those found at the intake. The circulation pattern inside the cove seems to influence the distribution of these pollutants in the sediments.

INTRODUCTION

The use of nearshore sites to locate electric power generation stations is quite common in Puerto Rico. The main advantage is the availability of the sea as both cooling water source and heated water discharge site. Once-through cooling systems are cost effective as compared to other options.

Corrosion and biofouling are common problems when sea water is used for cooling purposes. To reduce these problems to manageable levels, certain chemicals are used. Chromium in the form of chromate is used as corrosion inhibitor. Chlorine is used as antifouling agent.

In the cooling process, antifouling chemicals, corrosion inhibitors and heat are added to the intake water and discharged back to the sea. Petroleum hydrocarbons and other contaminants from the plant's processes are also commonly discharged with the heated sea water. Significant pollution of the receiving marine environment can result from leaching and erosion losses of metals from heat exchanger surfaces. These are often made of Ni/Cu alloys.

The effects of these pollutants in the marine environment is often overlooked when compared with the emphasis in the literature on research on the effects of heat itself.

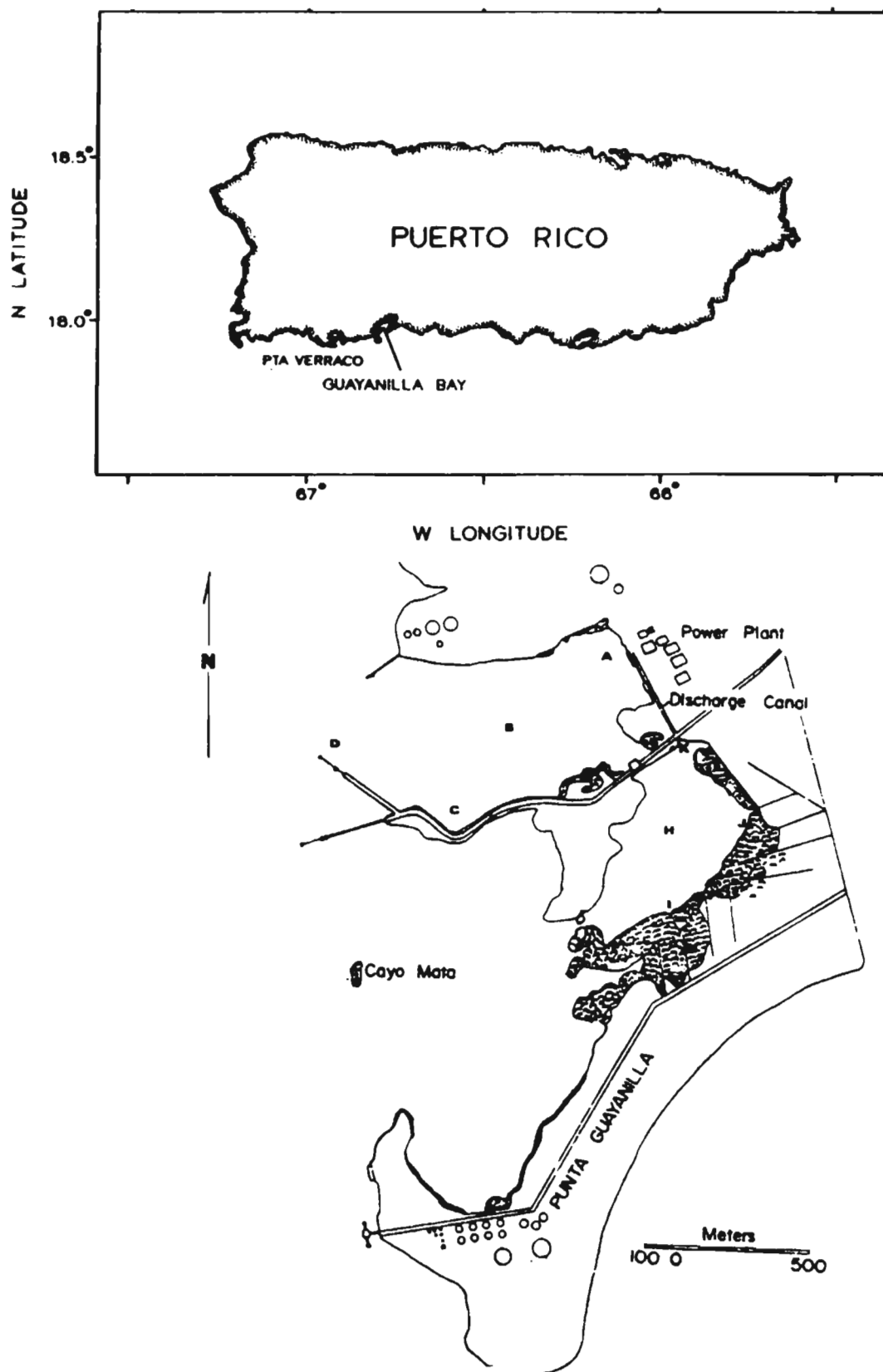
In this paper, we demonstrate the increased accumulation of selected trace metals and petroleum hydrocarbons in marine sediments of the discharge side, relative to the intake side, of an oil-fired power station using once-through cooling on the Caribbean Sea coast of Puerto Rico.

METHODS AND MATERIALS

Study Site

Bahía de Guayanilla is an embayment enclosed by coral reefs and lined by mangroves, which was the site of a large petrochemical industry complex located on the Caribbean Sea shore on the south coast of Puerto Rico (Figure 1). Studies of the physical, chemical and biological characteristics of the bay and of the overall impact of the industrial operations on this marine environment are summarized by Lopez (1979a).

Figure 1. Study site showing sampling station plan. Stations A,B,C,D are located in Intake area. Stations H,I,J,K are located in Receiving area (Thermal Cove).



Among the industries in the area, a petroleum refinery, various petrochemical manufacturing plants and the Costa Sur Power Generation station were major dischargers to the bay for many years. The oil-fired power station generates 1,135 MW and continues to discharge its once through-cooling water through a 100m canal into a nearly enclosed, mangrove lagoon (Thermal Cove), 900m long by 200m wide, which in turn is connected with the eastern portion of the bay by a 30m wide entrance. The high velocity discharge (>1 m/s) induces a clock-wise circulation within the Thermal Cove (Goldman, 1979).

Sampling and Analytical Methods

The samples were collected from R/V Sultana in 4 stations within the intake area and in 4 stations in the Thermal Cove (Figure 1). The sediment samples were obtained with a Shipek Dredge. An undisturbed portion (250 ml) of the samples (upper 5 cm) were removed and placed in plastic freezer containers for metal analysis and in glass jars for hydrocarbons analysis.

For trace metals analysis, portions of the sediments were oven dried overnight at 60°C to avoid metal losses. Then, the sample was ground to a fine powder in a porcelain mortar. Triplicate 1g subsamples were weighed into teflon beakers and digested with hydrogen peroxide, hydrofluoric acid and inverse aqua regia (3 parts HNO_3 : 1 part HCl). The filtered digests, made up to 25 ml with deionized water, were analyzed by direct aspiration, using D_2 background correction on a Perkin-Elmer 303 Atomic Absorption Spectrophotometer.

Petroleum hydrocarbons in the sediments were determined gravimetrically after Soxhlet extraction with benzene/methanol as per the method of Farrington and Tripp (1975).

RESULTS

The Cu, Ni, Zn and petroleum hydrocarbons concentrations in the sediments of the intake and Thermal Cove (discharge area) are given in Table 1. In all cases the pollutant concentrations at station H, which is located in the center of the cove, are higher than the concentration at the four stations within the intake area.

TABLE1. Pollutants concentrations in the surface sediments of Bahía de Guayanilla. Standard deviation in parenthesis.

STATION	HYDROCARBONS (%)	Ni (µg/g)	Zn (µg/g)	Cr (µg/g)	Cu (µg/g)
INTAKE					
A	0.19	20.51 (0.87)	55.26 (0.01)	29.86 (4.93)	50.75 (5.37)
B	0.04	16.08 (0.58)	57.99 (0.01)	71.24 (0.01)	67.91 (1.87)
C	0.14	11.99 (0.01)	58.72 (0.05)	26.74 (0.02)	75.45 (0.07)
D	0.05	14.49 (1.24)	67.00 (0.00)	75.21 (5.60)	78.96 (3.47)
THERMAL COVE					
H	0.28	47.74 (0.00)	90.98 (0.01)	122.7 (0.01)	117.3 (4.61)
I	0.25	15.49 (0.43)	66.97 (0.04)	38.92 (2.50)	36.48 (1.52)
J	0.34	52.21 (0.04)	34.17 (0.17)	22.32 (1.75)	104.2 (0.09)
K	0.15	18.56 (23.78)	24.07 (6.45)	8.35 (4.05)	5.86 (8.15)

A t-test was used to determine if the average concentration in the intake and the concentrations at the stations within the Thermal Cove were significantly different. Table 2 shows the results of the t-tests.

For Cu and Ni, stations H and J presented the highest concentration of all the stations samples (Figure 2). The average concentrations of these two stations are significantly different ($p \leq 0.5$) from the average concentration in the intake area. Stations I and K have Cu and Ni concentrations similar to those in the intake.

The average concentration of petroleum hydrocarbons in stations H, I and J is significantly higher than the average concentration in the intake area.

Station H has the highest Zn and Cr concentrations of all the stations sampled but the other stations in the Thermal Cove do not show enrichment relative to the intake. To compare this single observation with the mean of the intake area a special t-test was used (Sokal and Rohlf, 1981). The Cr and Zn concentrations at station H differ significantly ($p \leq 0.05$) from those in the intake area.

DISCUSSION AND CONCLUSIONS

The analysis presented demonstrates that a net accumulation of pollutants has occurred in the Thermal Cove relative to the intake area. We infer that this enrichment effect can be attributed to the power plant operation since there exists no other source of pollutant to the cove. The distribution of these pollutants in the sediments is the result of the circulation pattern inside the cove.

Station H is located at the center of a clock-wise gyre which is formed by the high speed (>1 m/s) thermal discharge. At the eastern end of the Thermal Cove, where station J is located, a small counterclock-wise gyre is also formed (Goldman, 1979).

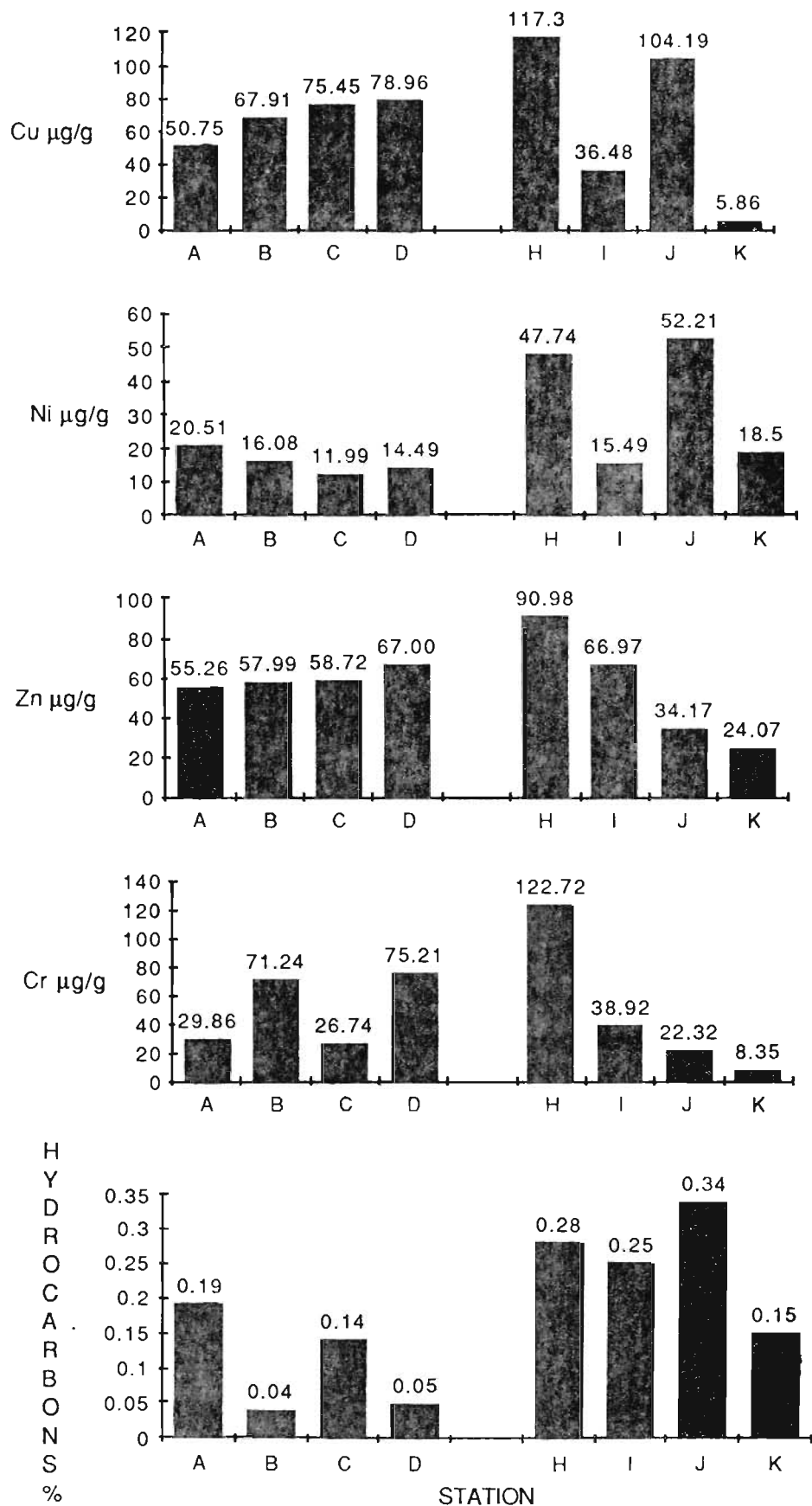
These two gyres appear to accumulate sediments and pollutants. In station K, at the outlet of the discharge canal, the high water velocity does not permit the accumulation of particles and pollutants.

Trace metals and petroleum hydrocarbons have been shown to be widely available to biota in the Bahía de Guayanilla and in similar environments. Bioaccumulations of trace metals in the mangrove oyster

Table 2. T-tests for comparing Intake and Thermal Cove Stations.

Cu in sediments ($\mu\text{g/g}$)	
Stations Compared	
All in Intake vs. H and J in Thermal Cove	
Average Intake = 68.3	Average Thermal Cove = 110.7
Standard deviation = 12.6	Standard deviation = 9.3
N = 4	N = 2
t(4) = -4.151	p = .015
Ni in sediments ($\mu\text{g/g}$)	
Stations Compared	
All in Intake vs. H and J in Thermal Cove	
Average Intake = 15.8	Average Thermal Cove = 49.97
Standard deviation = 3.6	Standard deviation = 3.16
N = 4	N = 2
t(4) = -11.334	p = .001
Petroleum Hydrocarbons in sediments (%)	
Stations Compared	
All in Intake vs. H, I and J in Thermal Cove	
Average Intake = 0.10	Average Thermal Cove = 0.29
Standard deviation = 0.07	Standard deviation = 0.05
N = 4	N = 3
t(5) = -3.83	p = .013
Cr in sediments ($\mu\text{g/g}$)	
Stations Compared	
All in Intake vs. H in Thermal Cove	
Average Intake = 50.76	Station H = 122.72
Standard deviation = 26.01	N = 1
N = 4	
t(3) = 2.50	.025 < p < .05
Zn in sediments ($\mu\text{g/g}$)	
Stations Compared	
All in Intake vs. H in Thermal Cove	
Average Intake = 59.74	Station H = 90.98
Standard deviation = 5.06	N = 1
N = 4	
t(3) = 6.90	p < .05

Figure 2. Pollutants concentrations in the surface sediments of Bahía de Guayanilla. Stations A to D were located in the intake, stations H to K were located in the Thermal Cove.



Crassostrea rhizophorae and in the flat tree oyster Isognomon alatus in Bahía de Guayanilla was demonstrated by Almodóvar-Ferrer (1986). Schroeder and Thorhaug (1978) and Thorhaug and Schroeder (1978) demonstrated the accumulation of heavy metals in the turtle grass Thalassia testudinum. Banus (1977) and López (1979b) showed incorporation of trace metals in the red mangrove Rhizophora mangle in the Bahía de Guayanilla area. These two plant species form the basis for important marine food webs in the Caribbean and Gulf Coast region. Biological magnification via these food webs may lead to humans.

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EFFLUENT TOXICITY TESTING FOR INDUSTRIAL DISCHARGERS

Mark Shearon
Harmon Engineering Associates, Inc.
Auburn, Alabama

ABSTRACT

The United States Environmental Protection Agency has developed final protocols for the testing of industrial and municipal wastewater discharges for toxicity to aquatic organisms. These testing procedures define the methods for estimating both the acute (immediate or short-term) toxicity, as well as the chronic (long-term) toxicity. Several organisms have been standardized, as well as the age and culture conditions for those organisms. These requirements define a necessary Quality Assurance plan for the toxicity laboratory. As applied by most state regulatory agencies, effluent toxicity is required of most primary and secondary industry. Some states initiated the acute testing within the past few years. Those dischargers currently obtaining renewed or new NPDES permits for process wastewater sources are having the testing added to, or maintained on, their discharge permits. The current approach to toxicity testing for NPDES permits involves a tiered approach to determining the required levels of testing. Initially, the effluent is "screened" to determine the acute toxicity. Those sources failing the screening test are then subjected to a more rigorous series of tests to define the LC₅₀, as well as the chronic toxicity values. Discharges which are consistently toxic are then required to enter into a Toxicity Reduction Study, wherein the toxic elements are identified, and additional treatment technology to correct the toxicity is evaluated. The end result is supposed to be a revision of the waste treatment technology to prevent further discharges of toxic effluents. This paper will identify the testing techniques currently required of the permittees, the QA requirements, the use of the results, and the apparent future of effluent toxicity testing.

TOXICITY AND BIOACCUMULATION TESTING

REVIEW PAPER

William Peltier
U.S. Environmental Protection Agency
College Station Road
Athens, Georgia 30613

Before I review the session papers on toxicity and bioaccumulation testing, I would like to say that, even though there were only four papers in this session, it does not diminish the importance of toxicity testing and bioaccumulation testing now or in the future. We are currently faced with the immense problem of toxic materials in our environment; the problem will only grow in the future. In addition to the historical chemical by chemical approach, the biological approach for the control of toxics has been accepted as an important tool in evaluating the impact of point and nonpoint source discharges to our environment.

Let me discuss what is currently happening in the areas of toxicity and bioaccumulation testing. Nationally, there is going to be a concerted effort to control toxics in the air, ground water, and surface water. In 1984 EPA recommended that biological monitoring, specifically toxicity testing requirements, be included in the NPDES permits. Since the most recent funding of the CERCLA (Superfund) and RCRA programs, bioassessment studies are being used to complement the chemical and physical approach in toxic control. Bioaccumulation studies are being used to determine the human risk associated with the consumption of fish and shellfish. We have gone through a process of encouraging regulatory permit writers and permittees to accept toxicity and bioaccumulation testing as a necessary means to control toxics. This acceptance has been a matter of education. Permit writers and permittees had to be educated as to how the data was to be interpreted and its application in a regulatory manner. Also for these programs to be implemented by the regulatory agencies, acceptable and cost-effective methods had to be available.

I use the term cost-effective because, as a biologist, I was probably my own worst enemy at planning elaborate studies. Early in my career, I was involved in many biological field monitoring studies. These studies were usually long-term, resource intense, and costly. At times the information was not provided in a timely manner so as to make a quality decision. As a result, other short-term test methods were employed to assess the toxic impact on the receiving water biota.

One of the first short-term test methods employed in the NPDES permit program was acute static and flow-through toxicity tests. These tests ranged from 24 to 96 hours in duration. Recently, short-term chronic toxicity tests have been developed and are in use for determining the toxicity of an effluent. These tests ranged from 1 hour to 9 days in duration.

The shift from field assessments to point source assessments of pollution raised a question as to the validity of the results of point source testing when compared to the traditional receiving water evaluations. As a result, EPA-ERL Duluth Laboratory conducted eight studies around the country to determine if end-of-the-pipe data correlated with field data. Each study involved whole effluent toxicity testing, receiving water dye studies, invertebrate and fisheries studies. In all cases, the results demonstrated that if toxicity occurred at the end-of-the-pipe, there was a similar impact on the receiving water biota.

North Carolina recently conducted similar comparisons using the short-term chronic test with Ceriodaphnia and evaluated the downstream impacts from point source discharges. They evaluated the macro-invertebrate population instream, and in 87% of the cases they were able to correlate "end-of-the-pipe" impacts with the instream impacts. Where there was no impact at the end-of-the-pipe based on the reproduction of Ceriodaphnia, there was no significant difference in the number of taxa found upstream vs. downstream. When there was no impact at the end of the pipe, there was a definite reduction in number of taxa found downstream from the discharge.

The short-term tests are not always a substitute for field studies. Field studies are important in determining long-term trends, baseline inventories or perturbation from episodic events. Selection of the

method to be used in assessing an impact on the biota must be on the case by case bases.

One of the problems in employing the short-term approach was who could do the testing for the permittees? To answer this question, EPA Region IV conducted a survey in 1986 on the number of consulting firms in the southeast that could do toxicity testing. As a result, approximately 36 consulting firms in Region IV could provide clients with some level of toxicity testing. The cost for these tests ranged as follows: 24 to 96 hours acute static tests on a single species range from \$50 to \$600; short-term chronic tests range from \$500 to \$2000. The costs appear to be negotiable based on the number of species to be tested and frequency of testing. The rationale for not putting toxicity testing requirements in the NPDES permits can't be justified based on lack of laboratories or cost. Certainly the costs are more reasonable when compared to earlier life cycle trout studies which required 2 years to complete (i.e., egg to egg studies).

The papers given during this session deal with once through cooling water, fly-ash effluent, bioaccumulation and industrial discharges. Each of which has a toxic impact on the receiving water biota. The paper titled "Pollutant Enrichment of Sediments from Cooling Water Discharge in Bahia de Guayanilla, Puerto Rico" brings out the fallacy that once through cooling water (i.e., non-contact cooling water) is free of pollutants. Most of the NPDES permits do not require the permittee to monitor for toxicity. As a result, periodic checks on selected cooling water discharge have indicated acute and/or chronic toxicity. Use of anti-corrosive and anti-fouling chemicals are now being closely monitored to insure toxic conditions are minimized or eliminated. Request for use of different chemical for anti-corrosive and fouling have to have data indicating the acute and chronic toxicity of the chemical to selected fish and invertebrates. The discharge of pollutants, even at low concentrations, poses a problem in the receiving water as indicated in the authors' paper. Concentrations of chemicals in the sediment may eventually reach levels that are toxic to the benthic organisms. The potential for bioaccumulation in the food chain increases as the area of contamination increases. The situation which is occurring in Puerto

Rico, certainly calls for some type of receiving water biological monitoring program. A reduction in the receiving water faunal community and/or an accumulation of pollutants in fish or shellfish would justify remedial action by the regulation agency. The second paper titled "Some Observations on Aquatic Impacts of Power Plant Trace Element Discharges" demonstrates the need for a greater concern of toxic effects from trace elements. The presents of trace elements in discharge from fly-ash settling ponds are usually at sublethal or chronic concentrations. For example, in Belews Lake selenium was discharged from the fly-ash settling ponds at ppb concentrations. However, the sediment concentrated selenium up to 50 ppm which then became a source of toxicity to the biota. Selenium was then bioconcentrated in the tissues of fish through the food chain. As a result, selected populations of fish were eventually eliminated through a chronically toxic condition. Species sensitivity to selenium was also demonstrated by the fact that the fathead minnow was not affected; however, native Centrarchids and Ictalurids were eliminated. The point was made that further research is necessary to answer the questions regarding the toxic impact of trace elements.

The bioaccumulation of radionuclide in a predator was illustrated in the paper titled, "Radiocesium Kinetics in the Yellow-Bellied Turtle (Pseudemys scripta). Resident biota, such as certain turtles and raptors, feed on contaminated food sources. At the Savannah River Plant, cooling ponds, such as Parr Pond, have a fish community that is contaminated by ^{137}CS . The contamination probably comes from both the ^{137}CS in the water column and through the food chain. The contaminated fish are sources of the bioaccumulation in predator organisms. The kinetics involved with uptake and depuration rates with many predator organisms is not well documented. Results from this paper does shed some light on the utility of modeling the information, however much work still needs to be done in the future.

The paper on "Effluent Toxicity Testing for Industrial Dischargers" covered much of my early discussion on current activities in toxicity testing. However, if effluent is consistently toxic, the permittee would be required to conduct a Toxicity Reduction Evaluation (TRE). The purpose of the TRE is to identify the cause of toxicity and determine

treatment necessary to reduce or eliminate the toxicity. The TRE is divided into three phases: Phase I - Toxicity characterization; Phase II - Causative toxicant identification; and Phase III - Causative toxicant confirmation. What is unique in the TRE is the use of an aquatic organism i.e., daphnid, as the analytical detector of toxicity. Acute toxicity test are employed throughout all three phases. Minimal analytical work is required in Phase I, as the majority of the identification is conducted with toxicity tests. Phases II and III may require extensive analytical support.

As indicated from the papers presented in this section, the area of toxics and bioaccumulation is going to be in the forefront of pollution abatement for years to come.

SOUTHEASTERN WORKSHOP ON
AQUATIC ECOLOGICAL EFFECTS OF POWER GENERATION

December 3-5, 1986

Mote Marine Laboratory
Sarasota, Florida 33577

POSTER SESSION ABSTRACTS
(Arranged Alphabetically)

1. Bruzek, D.A. and S. Mahadevan. Mote Marine Laboratory, Sarasota, Florida.
THE APPLICATION OF LOG-NORMAL DISTRIBUTIONS IN DETECTING POLLUTION-INDUCED CHANGES IN A BENTHIC COMMUNITY.
Benthic infaunal samples were collected from 40 stations over a period of 15 months in the vicinity of the Crystal River Power Station. The individuals of each species were identified and enumerated, and the log-normal distribution of individuals per species was determined. Curves were drawn for each station and sampling period and mean angles calculated from these curves. The mean angle of each station was then plotted as a means of detecting the pollution-induced changes as well as naturally disturbed areas in the communities. It was found that stations in the area of the thermal effluent and naturally disturbed areas had lower log-normal angles (30-35°), whereas the offshore and nearshore undisturbed stations had higher log-normal angles (above 40°). Since a shallower slope reflects a community under environmental stress, these data suggest that the alteration of basic log-normal distribution of benthic communities can be used to discern disturbances.

2. Frere, P. Potomac Electric Company, Washington, DC.
CONTROL OF MACROFOULING ON INTAKE STRUCTURES AND MODIFICATIONS TO CHLORINATION PROCEDURES AT AN ESTUARINE POWER PLANT.
Potomac Electric Power Company's Chalk Point Station is located on an estuarine segment of the Patuxent River, a tributary to the mid-Chesapeake Bay. The station has two open-cycle, 355 MWe units using condenser cooling water taken directly from the Patuxent River. The cooling water was previously continuously chlorinated to control macrofouling organisms. Over the years, the station has suffered numerous unit outages due to biofouling of the intake gallery and precondenser piping surfaces which are not exposed to chlorine. Dense mats of Garveia sp. accumulate on concrete surfaces during the summer months. These mats serve as habitat for secondary fouling organisms. The mats periodically break free from the walls and block the condenser tube sheet face, reducing cooling water flow and causing unit shutdowns. A study to solve the condenser tube blockage problem by documenting the seasonal growth curve for Garveia, relating the growth to natural cycles of water temperature and salinity, and identifying secondary fouling organisms was initiated in 1985. Concrete plates similar in surface characteristics to the intake structure were suspended in front of the station intake screens and allowed to foul. The plates were

inspected at intervals through the period July 1985 to October 1986. The study included an evaluation of various antifouling coatings to control growth of Garveia. Plates similar to those employed in the monitoring program were primed and painted with Intersleek, a non-toxic ultra-smooth material, and No Foul, a paint containing tributyltin oxide. These plates were suspended in the intake along with unpainted controls and inspected at weekly intervals through the early summer fouling period. Growth of Garveia and secondary fouling organisms was recorded. Intermittent chlorination was attempted in 1986 to reduce the rate of condenser tube corrosion. Records were kept on condenser backpressure, circulating water pump amperage and chlorine dosage. Inspections of intake condenser waterboxes were made during each outage over the test period.

3. Graham, R.J. North Anna Environmental Lab, Virginia Power, Virginia.

FACTORS AFFECTING THE SUCCESS OF ARTIFICIAL FISH STRUCTURES IN A POWER STATION'S COOLING RESERVOIR.

Clear-cutting of timber and removal of man-made structures are common practices in the construction of impoundments and reservoirs used for cooling facilities. These activities may result in limited habitat for cover-oriented fishes, e.g., centrarchids and ictalurids, and reduce the probability of developing productive fisheries. Lake Anna, Virginia is a 24,000 hectare impoundment of the North Anna River that provides cooling water for Virginia Power's North Anna Power Station. The lake site was clear-cut prior to construction, and approximately 90% of the lake bottom consists of sand, silt or clay. Seven artificial fish structures were placed in Lake Anna by Virginia Power during 1983 at depths ranging from 5 to 9 m in an attempt to provide cover for the sport species largemouth bass, Micropterus salmoides, and black crappie, Pomoxis nigromaculatus, nursery areas for a major forage species, bluegill, Lepomis macrochirus, and fishermen with opportunities to concentrate their efforts in areas easily accessible and fished by traditional methods. Based on SCUBA observations and water quality monitoring surveys, factors that appear to have significantly influenced utilization of fish structures include the construction materials used and pumping operations of the North Anna Power Station. Structures consisting of cedar trees anchored over cinder block substrates were generally unproductive, due to accretions of silt that compacted boughs, in comparison to structures consisting of hardwood tree tops anchored over cinder block substrates. Structures placed in areas subject to currents resulting from pumping operations exhibited more acceptable (D.O. 5 ppm) oxygen levels during the summer months than those placed in areas protected from or out of the influence of pumping operations, due to better mixing of well oxygenated epilimnetic and poorly oxygenated hypolimnetic waters. Important considerations in construction and location of fish structures in cooling facilities should include clear definition of results desired, choice of target species, availability and cost of construction materials, temperature and oxygen requirements of target species and hydrologic characteristics of the waters.

4. **Grimes, D.V.** North Anna Environmental Lab, Virginia Power, Virginia.
A MOUNTED HYDROPHONE BRACE FOR USE IN BIOTELEMETRY STUDIES.
Biotelemetry studies have traditionally been carried out with the use of hand-held hydrophones. In 1984 a two year biotelemetry study of striped bass, Morone saxatilis, was begun at Lake Anna, Virginia, which serves as a 24,000 hectare cooling reservoir for Virginia Power's North Anna Power Station. The need for a more effective and efficient means of locating and tracking tagged fish led to the development of a boat mounted hydrophone brace. A boat mounted hydrophone offers several advantages over hand-held hydrophones. Advantages include the reduction of manpower requirements for tracking surveys by 50%, and the capability to easily collect large amounts of accurate tracking data as a result of being able to move with located fish. The mounted hydrophone brace has been used at Lake Anna since mid 1984. During the period 1984-1986, 73% of the fish tagged have been located and tracked using the boat mounted hydrophone; 63% of the tags found were tracked for 1-3 months, 34% were tracked for 4-6 months, and 3% were tracked for more than one year. The cost of the hydrophone brace was under thirty dollars.
5. **Horst, T.J., J.K. Downing, T.A. Biffar.** Stone & Webster Engineering Corporation, Boston, Massachusetts.
ENTRAINMENT AS A RISK ASSESSMENT PROBLEM.
Entrainment of organisms with intake water has received much attention, and the projected impacts have resulted in the expenditure of considerable economic resources in an attempt to reduce entrainment. Related decisions have necessarily been made in light of large uncertainties with respect to the ecological effects of entrainment and the effectiveness of technologies to mitigate entrainment. One method for investigating these decisions is risk assessment, which has received wide application in fields as diverse as engineering economics and environmental health analysis. In risk analysis the risk is determined by both the exposure to the risk source and the consequence. Entrainment at water intakes is examined as a risk assessment problem. Alternative mitigative measures are examined with respect to risk reduction. The problem is formulated both from the environmental point of view, risk to the organism, and from the economic point of view, risk of capital. Selected site data are used for illustrative purposes.
6. **Loeffelman, P.H. and J.B. Suomala.** American Electric Power Service Corporation, Columbus, Ohio.
UNDERWATER ACOUSTIC RADIATION, WATER VELOCITY AND WARMWATER FISH DISTRIBUTION AT RACINE HYDROELECTRIC PROJECT, USA.
Underwater acoustic radiation, water velocity and hydroacoustic (SONAR) fish monitoring was conducted in August 1986 at Ohio Power Company's 48,000 kW Racine Hydroelectric Project at Ohio River mile 237.5 near Racine, Ohio, USA. This monitoring was performed to determine if any relationships could be established among: a) underwater sound produced by the two bulb (horizontal Kaplan) turbines; b) water velocity; and c) presence of warmwater fish species in the forebay, intake, draft tube, and tailwaters of the project. Acoustic radiation and water velocity measurements at 50

locations were conducted, while visual and SONAR observations of fish distributions were made at a number of these locations. Total and mean RMS pressure levels of spectra from 2 Hz to 20 kHz were measured. Highest pressures were within the probable auditory range (50 to 800 Hz) of resident teleost fish. Of all locations, continuous sound pressure levels from the bulb turbines were highest in the draft tube (+158 dB//1 uPa) and immediately upstream of the intake trashracks (+185 dB//1 uPa). The pressure level of +138 dB//1 uPa in the tailrace, the lowest level recorded at all locations, was 139 times lower than the highest level recorded. High acoustic intensity, discrete narrow band frequency lines at 120, 240, 360 and 720 Hz were also recorded. Fish were observed at all water velocities, including those as high as 6 fps in the tailrace. Fish were unevenly distributed among monitoring locations. Few fish were observed in the draft tube and immediately upstream of the intake trashracks compared to most other locations. The monitoring information suggests that differences in the level of acoustic radiation contribute to different fish distributions.

7. **Massie, F. Virginia Power Company, Virginia.**
IMPACT MITIGATION AND ENVIRONMENTAL ENHANCEMENT AT VIRGINIA POWER'S BATH COUNTY PUMPED STORAGE PROJECT.

The Bath County Pumped Storage Project is the largest hydroelectric facility of its type in the world. Construction was completed, and the plant began commercial operation in 1985. The project, located in the western mountains of Virginia, is jointly owned by Virginia Power and the Alleghany Power System. Construction of the lower reservoir for the project inundated approximately 3.3 miles of Back Creek, a third order valley stream. To mitigate for this loss, Virginia Power proposed a three-fold plan which was accepted by the Federal Energy Regulatory Commission, which included: 1) construction of a public recreation area; 2) a stream habitat improvement plan; and 3) a wild turkey habitat enrichment program. The public recreation area was designed by the Harza Engineering Company and constructed by Virginia Power. It includes two stocked ponds (approximately 54 and 27 acres) for both bank and boat fishing, campgrounds, picnicking areas, a public beach, swimming area, and a bath house. Virginia Power biologists will monitor the fishery in conjunction with representatives from the Virginia Commission of Game and Inland Fisheries. The Stream Improvement Plan was designed by Dr. John Ney of Virginia Polytechnic Institute and State University to improve the fish habitat in a 1.5 mile section of Back Creek located below the lower reservoir of the project. The design called for the development of pools, stable banks and instream cover. Construction of the improvement area was completed in 1985. Biological monitoring of the area is being conducted by representatives of VPI for two years following construction to assess the effectiveness of the modifications. Virginia Power will then continue monitoring the area for the following five years. Small clearings were made at strategic locations around the project site for planting of vegetation as a food source for wild turkeys. These clearings have been very successful in attracting turkeys. They are "maintained" periodically to ensure their effectiveness.

8. Marion, W.R. University of Florida. J.R. Wilcox. Florida Power & Light Co.
BALD EAGLE PRODUCTIVITY NEAR FLORIDA POWER & LIGHT COMPANY FACILITIES.
The majority of Southern Bald Eagle nesting occurs in Florida, with 220-250 successful nests each year. Florida Power & Light Company conducted a long-term monitoring program at selected Southern Bald Eagle nests in central Florida in an attempt to anticipate and minimize potential disturbances during the nesting season. Productivity at four such nesting locations is compared with statewide productivity averages for 1974-1978.
9. Mazzotti, F.J. University of Pennsylvania. J.R. Wilcox. Florida Power & Light Company.
CROCODILE MONITORING AT FLORIDA POWER & LIGHT COMPANY TURKEY POINT PLANT.
Florida Power & Light Company's crocodile monitoring program has documented the compatibility of this endangered species with the generating of electricity. Over the last ten years, monitoring has determined the number of nests, growth and survival of hatchlings, growth and survival of juveniles, and the abundance and distribution of adults. Two nests hatched each year during 1984-1985 with a total of 55 hatchlings being tagged. Twelve of the 34 hatchlings tagged in 1984 were recaptured in 1985. Sixteen juveniles were captured over the monitoring period, and seven were recaptured at least once. Crocodiles continue to nest, grow, and survive at the Turkey Point Power Plant site.
10. Sprinkel, J.M. and J.F. Gorzelany. Mote Marine Laboratory, Sarasota, Florida.
THE EFFECTS OF THERMAL EFFLUENT ON OYSTERS (*Crassostrea virginica*) AND OYSTER REEF FAUNA IN THE VICINITY OF FLORIDA POWER CORPORATION'S CYRSTAL RIVER POWER PLANT.
The height, length, weight, and volumetric displacement of oysters were measured before and after deployment in control and thermal areas. Mortality rates were noted, and a Condition Index (CI) was determined for each live oyster returned. Growth in control areas was not significantly different than growth in areas of maximum thermal influence. Growth was enhanced, however, in areas of moderate thermal influence. CI correlated with growth and spawning, while mortality rates were highest in late summer and in the immediate thermal area. A total of 175 taxa of oyster associated fauna were identified and enumerated. Abundances and diversity of oyster associated fauna decreased in thermally affected areas, along with a change in community composition from a mollusc-dominant community, to a more polychaete-dominant community. This alteration was not widespread, however, and was essentially limited to areas undergoing immediate thermal influence.

**SOUTHEASTERN WORKSHOP ON AQUATIC ECOLOGICAL
EFFECTS OF POWER GENERATION**

LIST OF PARTICIPANTS

Rafael Almodovar-Ferrer
University of Puerto Rico
Center for Energy and Environment Research
College Station
Mayaguez, Puerto Rico 00708
(809) 832-7912

John Alonso
CH2M Hill, Inc.
3030 North Rocky Point Drive West, Suite 350
Tampa, Florida 33607
(813) 888-6777

A. Spencer Autry
Tampa Electric Company
Post Office Box 111
Tampa, Florida 33601
(813) 228-4838

Jeffrey O. Barnes
Barnes-Williams Environmental Consultants
132 Washington Street
Binghamton, New York 13901
(607) 723-3113

Paul J. Behrens
Florida Power Corporation
3201 34th Street South
St. Petersburg, Florida 33711
(813) 866-5521

Chris Benedict
Carolina Power & Light Company
Brunswick Biology Laboratory
Post Office Box 10429
Southport, North Carolina 28461
(919) 457-3397

Dr. Thomas Biffar
Stone & Webster Engineering Corporation
245 Summer Street
Post Office Box 2325
Boston, Massachusetts 02107
(617) 589-2725

LIST OF PARTICIPANTS continued.

James D. Brown
U.S. Fish and Wildlife Service
75 Spring Street Southwest
Suite 1276
Atlanta, Georgia 30303
(404) 331-6343

Vilma S. Brueggemeyer
Tampa Electric Company
Post Office Box 111
Tampa, Florida 33601
(813) 228-4841

David A. Bruzek
Mote Marine Laboratory
1600 City Island Park
Sarasota, Florida 33577
(813) 388-4441

John P. Buchanan
Tennessee Valley Authority
Fish & Aquatic Ecology Branch
145 Summer Place Building
Knoxville, Tennessee 37902
(615) 632-3701

Linda R. Cadman
Martin Marietta Environmental Systems
9200 Rumsey Road
Columbia, Maryland 21045
(301) 964-9200

John Christian
U.S. Fish & Wildlife
75 Spring Street Southwest
Atlanta, Georgia 30303
(404) 331-3580

Dr. Sheree L. Cohn
St. Cloud State University
Department of Biological Sciences
St. Cloud, Minnesota 56301
(612) 255-4912/253-8634

Peter M. Cumbie
Duke Power Company
Design Engineering Department
Post Office Box 33189
Charlotte, North Carolina 28242
(704) 373-8331

LIST OF PARTICIPANTS continued.

Robert M. Daniels
Virginia Power
North Anna Environmental Laboratory
Post Office Box 402
Mineral, Virginia 23117
(703) 894-5151

Dr. William Davies
Department of Fisheries Allied Aquacultures
Auburn University, 203 Swingle Hall
Auburn, Alabama 36849
(205) 826-4786

Robert W. Davis
Lawler Matusky and Skelly
Post Office Box 1509
One Blue Hill Plaza
Pearl River, New York 10965
(914) 735-8300

Kathleen Durrell
Tampa Electric Company
Post Office Box 111
Tampa, Florida 33601
(813) 228-4842

Bruce W. Easley
Virginia Power
Surry Power Station
Post Office Box 315
Surry, Virginia 23883
(804) 357-3184

Robert G. Ernest
Applied Biology, Inc.
Post Office Box 974
1458 Sunview Terrace
Jensen Beach, Florida 33457
(305) 334-3729

Phillis E. Frere
Potomac Electric Power Co.
Hallowing Point Lab
RR 2, Box 81
Prince Frederick, Maryland 20678
(301) 855-1295

Jay Gorzelany
Mote Marine Laboratory
1600 City Island Park
Sarasota, Florida 33577
(813) 388-4441

LIST OF PARTICIPANTS continued.

Robert J. Graham
Virginia Power
North Anna Environmental Laboratory
Post Office Box 402
Mineral, Virginia 23117
(703) 894-5151

Paul Graybill
Florida Power & Light/Mote Marine Lab
4111 Plumosa Terrace
Bradenton, Florida 33507
(813) 792-7956

David V. Grimes
Virginia Power
North Anna Environmental Laboratory
Post Office Box 402
Mineral, Virginia 23117
(703) 894-5151

Pete Hofmann
Stone & Webster Engineering Corp.
Post Office Box 2325
245 Summer Street
Boston, Massachusetts 02107
(617) 589-2841

Don Holmes
CH2M Hill
3030 North Rocky Point Drive
Tampa, Florida 33607
(813) 888-6777

T.J. Horst
Stone & Webster Engineering Corp.
245 Summer Street
Post Office Box 2325
Boston, Massachusetts 02107
(617) 589-2725

Charles H. Kaplan
U.S. Environmental Protection Agency
345 Courtland Street NE
Atlanta, Georgia 30365
(404) 347-3012

James Kelly
Pacific Gas & Electric Co.
Post Office Box 117
Avila Beach, California 93424
(805) 595-7193

LIST OF PARTICIPANTS continued.

Roy R. Lewis, III
Mangrove Systems, Inc.
Post Office Box 290197
7729 Professional Place
Tampa, Florida 33687
(813) 989-3431

Paul H. Loeffelman
American Electric Power Service Corp.
Environmental Engineering Division
One Riverside Plaza
Columbus, Ohio 43216
(614) 223-1243

Elaine R. Macinski
Florida Power Corporation
Post Office Box 14042
St. Petersburg, Florida 33733
(813) 866-4481

Dr. Kumar Mahadevan
Mote Marine Laboratory
1600 City Island Park
Sarasota, Florida 33577
(813) 388-4441

B.M. Marshall
Virginia Power
Post Office Box 26666
Richmond, Virginia 23261
(804) 775-5199

R. Erik Martin
Applied Biology, Inc.
Post Office Box 974
Jensen Beach, Florida 33457
(305) 334-3729

Frank Massie
Virginia Power
2043 Jackson Shop Road
Goochland, Virginia 23063
(804) 257-4779

Jack Mattice
Electric Power Research Institute
3412 Hillview Avenue
Palo Alto, California 94304
(415) 855-2763

LIST OF PARTICIPANTS continued.

Robert A. Mattson
Mangrove Systems, Inc.
Post Office Box 290197
7729 Professional Place
Tampa, Florida 33687
(813) 989-3431

Dr. Richard G. Monzingo
Commonwealth Edison Company
2 North LaSalle Street
Post Office Box 767
Chicago, Illinois 60690
(312) 294-4446

John M. Nestler
U.S. Army Engineering Waterways Experiment Station
Post Office Box 631
Vicksburg, Mississippi 39180
(601) 634-3870

Dr. Michael C. Newman
Savannah River Ecology Lab
Drawer E
Aiken, South Carolina 29801
(803) 725-2427

Dr. Lawrence Olsen
Florida Department of Environmental Regulation
Biology Section
2600 Blair Stone Road
Tallahassee, Florida 32301
(904) 487-2247

Robert G. Otto
R.G. Otto & Associates
620 Greene Street
Key West, Florida 33040
(305) 296-5838

William Peltier
U.S. Environmental Protection Agency
College Station Road
Athens, Georgia 30605
(404) 546-2294

Eric L. Peters
University of Georgia
Department of Zoology
Room 724, Box 23
Biosciences Building
Athens, Georgia 30602
(404) 542-3310

LIST OF PARTICIPANTS continued.

Cindy M. Reigel
Tampa Electric Company
Post Office Box 111
Tampa, Florida 33601
(813) 228-4834

Dr. John E. Reynolds, III
Eckerd College
Post Office Box 12560
St. Petersburg, Florida 33733
(813) 867-1166

Mike Roddy
Seminole Electric Cooperative, Inc.
Post Office Box 272000
Tampa, Florida 33688
(813) 963-0994

Dr. Roger A. Rulifson
East Carolina University
Institute for Coastal and Marine Resources
Greenville, North Carolina 27858
(919) 757-6220/757-6770

Paul M. Sawyko
Rochester Gas & Electric Corporation
89 East Avenue
Rochester, New York 14649
(716) 724-8399

Mark Shearon
Harmon Engineering Associates, Inc.
1550 Pumphrey Avenue
Auburn, Alabama 36830
(205) 821-9250

Jay Sprinkel
Mote Marine Laboratory
1600 City Island Park
Sarasota, Florida 33577
(813) 388-4441

John Suomala
50 Garrison Road
Bingham, Massachusetts 02043
(617) 749-7049

LIST OF PARTICIPANTS continued.

Edward P. Taft
Stone & Webster Engineering Corp.
Post Office Box 2325
245 Summer Street
Boston, Massachusetts 02107
(617) 489-2948

Bradley L. Weigle
Florida Department of Natural Resources
Bureau of Marine Research
100 8th Avenue SE
St. Petersburg, Florida 33701
(813) 896-8626

Dr. Ross Wilcox
Florida Power & Light Co.
Post Office Box 14000
700 Universe Boulevard
Juno Beach, Florida 33408
(305) 863-3623

Jerry L. Williams
Tampa Electric Company
Post Office Box 111
Tampa, Florida 33601
(813) 228-4837

Howard D. Zeller
California Department of Natural Resources
Floyd Towers East, Suite 1252
205 Butler Street, SE
Atlanta, Georgia 30334
(404) 656-2795

Charles J. Zimmerman, Jr.
Dames & Moore
455 East Paces Ferry Road
Suite 200
Atlanta, Georgia 30363
(404) 262-2915